The Importance of Life Cycle Assessment Methodology in the Regulation of Biofuels

Whittaker, Carly

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Carly Whittaker

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Abstract

Biofuels have been identified as a potential short-term solution for reducing greenhouse gas (GHG) emissions from road transport. In order to ensure that they successfully deliver emission savings, the overall GHG balance of producing them must be calculated accurately, and compared with conventional fossil fuels. Life cycle assessment has dominated the process of assessing the GHG emissions from biofuels, though the results can vary significantly, due to real variation caused by biomass feedstock types, processing stages, and uncertainty in GHG emission from certain processes, but also due to how the GHG emission balance is calculated.

This study has examined the relative importance of 'scientific' variation, and that caused by different methodological approaches. Three case studies with different methodological issues associated with accounting for their GHG emissions were developed. The variation in the LCA results due to the variability of inputs and outputs and from uncertainty of emissions from certain processes were assessed. The results showed that there is a high amount of variation in GHG emissions from fertiliser use, nitrous oxide emissions from soil and direct land use change.

The GHG emissions from the full bioethanol production system were then calculated according to three specific GHG regulatory methodologies that are either currently used, or have been used in the UK for biofuels, products and services. This study has found that LCA methodology can cause considerable variation in differences in LCA results. The variation is sometimes caused by arbitrary decisions concerning how GHG emissions from a process should be accounted for and attributed to the main product. Variation in the results that is caused by methodology is comparable to that caused by scientific variation, which is 'real' that sometimes cannot be avoided without detailed study.

The different results are due to the approach the methodology takes to LCA; whether the method tends toward attributional or consequential LCA. For reporting purposes, the European Directive's Renewable Energy Directive (RED) states that attributional LCA (ALCA) is best as it provides a snapshot of emissions that are released, and attributable to the production and use of the product or service. A consequential LCA (CLCA), on the other hand is better suited for policy analysis as the potential impacts are applicable to a wider, even global scope. None of the methodologies studied completely adheres to ACLA or CLCA. It appears that they have confused the two within their calculation rules; therefore they do not fulfil their goal and scope.

A critical assessment of the accounting methodology within RED is also performed here. The results show that the methodology is inconsistent and arbitrary, and currently too vague to be practical for GHG reporting. The results from this study indicate that the RED penalises the use of renewable energy and its calculation methodology does not support 2nd generation bioethanol production.

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List of Publications

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Glossary and Abbreviations

ALCA	Attributional life cycle assessment - provides information about the direct GHG emissions that are directly attributed to the production and use of a product (Sanchez et al., 2012)
Allocation	The process whereby upstream environmental impacts are split between co-products
BAT	Best available technology
BSI	British Standards Institute
By-products	Production residues that are not a waste
CH_4	Methane
CHP	Combined heat and power
CLCA	Consequential life cycle assessment - examines the GHG emissions that occur due to a change in production of a product (Brander et al. 2009a)
CO_2	Carbon dioxide
CO_2 eq.	Carbon dioxide equivalent
Co-products	Any of two or more products coming from the same unit process or product system
DALY	Disability-adjusted life year
DEFRA	Department of the Environment and Rural Affairs
d.f	Degrees of freedom
DDGS	Dry distillers grains and solubles
DLUC	Direct land use change
DNDC	DeNitrification and DeComposition model
Emission Factors	These are based on LCA studies in themselves, of which the results are stored in LCA databases, such as Ecoinvent, which is a peer-reviewed database (EcoInvent, 2007). These databases can provide a short-cut to estimating the GHG emissions from a process
FIE	Fertiliser induced emission
GHG	Greenhouse gas
GJ	Giga joule
GVW	Gross vehicle weight
ILUC	Indirect land use change
Impact assessment	Establishes a relationship between the product and its impacts on the environment
IPCC	International Panel on Climate Change
ISO	International Standards Organisation
K_2O	Potassium
HHV	Higher heating value
hp	Horse power

LCA	Life cycle assessment
LHV	Lower heating value
LUC	Land use change
MJ	Mega joule
N	Nitrogen
N_2O	Nitrous oxide
N_2O-N	Component of N in N_2O
N_{AG}	Nitrogen content above ground
N_{BG}	Nitrogen content below ground
NNFCC	National Non-Food Crops Centre
NREL	National Renewable Energy Laboratory
P_2O_5	Phosphorous
PAS2050	Publicly available standard 2050
pdf	Probability density function
PJ	Peta joule
RED	Renewable Energy Directive
RFA	Renewable Fuel Agency
RFS	Renewable Fuel Standard
RTFO	Renewable Transport Fuel Obligation
Shadow price	An estimated price of a product at the point of creation, minus any costs of further processing to the point of sale.
SD	Standard deviation
SOC	Soil organic carbon, a result of the long-term storage of atmospheric CO_2 as a relatively inert form of carbon with a potential residence time of decades to centuries (Kochsiek and Knops, 2012; Lal, 2008a)
SRC	Short rotation coppice
Substitution credit	A credit that is awarded for avoided production of a marginal good
System boundaries	Should clearly define the stages of a product's lifecycle and sources of environmental impacts that are accounted for
UNFCCC	United Nations Framework Convention on Climate change
Wastes	Any substance or object which the holder discards or intends or is required to discard
Zgi	"Z" Statistic for skewness and kurtosis.

Chapter 1. Introduction

Concern over world-wide climate change has led to an increased interest in identifying major sources and sinks of carbon and greenhouse gases (GHG). The UK is committed to providing annual GHG reports to the United Nations Framework Convention on Climate change (UNFCCC) and European Union (EU) as part of its legally binding Climate Change Act, committing it to reduce total National GHG emissions by 80% by 2050, using 1990 emissions as a baseline (HM Government, 2007). The Climate Change Act has 'galvanised' interest in assessing the sustainability of all sectors of the economy (Gadema and Oglethorpe, 2011). National-level reporting involves assessing sources and sinks of emissions from the energy, transportation, agriculture, forestry sectors and identifying land use changes that have occurred over time. Methodologies for GHG accounting on a National level have been developed through a widespread scientific panel of experts in the UNFCCC and International Panel on Climate Change (IPCC) and in parallel to this, there has also been a number of GHG reporting methodologies and tools developed to assess emissions occurring from the food and renewable energy sectors.

There is a need to reduce GHG emissions from road transport, as they represent 20% of total UK GHG emissions (DECC, 2011a) and is one of the only sectors where GHG emissions have increased since 1990, where sheer increases in kilometres travelled have overcome any emission savings achieved by adoption of efficient fuels and vehicle efficiency improvements (EEA, 2012). Although there are lower carbon options, with new vehicles adhering to stricter emission standards, as well as hybrid and electric vehicles, it has been estimated that a complete replacement of the vehicle fleet would require up to 16 years (WRAP, 2002), during which we may see alternative sustainable sources become available, such as hydrogen fuel cell technology (Royal Society, 2008).

It has been suggested that biofuels offer a short-term solution to reducing both demand for liquid fossil fuels and GHG emissions from the transportation sector (DTI, 2007). They can be used, at specified blends, in current car models, and a distribution network of liquid fuels already exists (Royal Society, 2008).

1.1. Assessing the GHG Balance of Biofuels

In order to ensure that GHG emissions are actually reduced by biofuels, it is vital that the GHG balance of producing and delivering them is favourable (Black et al., 2011). How this is measured is not only a serious issue but a complex one, as the emissions from a biofuel supply chain cannot be directly measured but require models of LCA to calculate the implications of biofuel production (Aylott et al., 2012).

It has been demonstrated that biomass is not 'carbon neutral' as previously thought (DTI et al., 2003). Bioenergy supply chains require the consumption of non-renewable resources to produce, transport and process them, and there may be negative impacts on converting land to

arable use (Royal Society, 2008; Searchinger et al., 2008). Therefore, if biofuels are made from plant material, it does not necessarily guarantee they have a lower emissions profile than conventional fossil fuels. It was realised that each biofuel, or biomass, from each feedstock, must be considered according to its own merits, and alongside specified sustainability criteria (Royal Society, 2008). It is important to identify under what conditions would a biofuel not reach its GHG emission savings targets, and determine how certain we can be of the emissions that occur during their production.

The increasing need to assess the environmental impacts from products and services has prompted the development of methodologies that account for impacts in a holistic way. Life cycle assessment (LCA) is a technique which has dominated this area of environmental impact assessment as it systematically accounts for all the impacts that arise during the production, use and disposal of a product (Plassmann et al., 2010). It is considered to be an adequate instrument for environmental decision support (Von Blottnitz and Curran, 2007).

A number of LCA studies have been performed on bioethanol production. Studies range from examining different production scenarios (EUCAR et al., 2006; Martinez-Hernandez et al., 2013; Punter et al., 2004; Weinberg and Kaltschmitt, 2013), location (Ahlgren et al., 2012; Webbs et al., 2010), feedstocks (Mortimer et al., 2004; Rosenberger et al., 2001), and effects of LCA modelling procedures (Gnansounou et al., 2008, 2009; Kaufman et al., 2010; Malça and Freire, 2006; Mendoza et al., 2008) or scales of production (Bernesson et al., 2006). It is reported that the majority of studies report that wheat and lignocellulosic bioethanol give favourable GHG emission results (Menichetti and Otto, 2008; von Blottnitz and Curran, 2007), however there are still a large range in the observed savings. Determining the reason for such variation in the results will help identify options for reducing uncertainty in the possible benefits of biofuels.

1.2. Problem Statement

In the literature, the results of LCA studies are based on specific case studies, contain specific assumptions and adopt different calculation methodologies, making direct comparisons complicated (Larson, 2006; von Blottnitz and Curran, 2007; Whitaker et al., 2010; Woods and Bauen, 2003). Behind the results of every LCA study is a myriad of decisions on the choice of analysis, methods and assumptions on material flows and uses (Aylott et al., 2012). More recent studies have focussed on examining why there is variation in LCA results between studies, and identifying uncertain sources of GHG emissions in GHG assessments of biofuels (Ahlgren et al. 2012; Rowe et al. 2011; Wang et al. 2012; Whitaker et al. 2010; Yan & Boies 2013).

Three major causes of variation of LCA results are identified:

1. ***Real variation*** – due to real differences in supply chains
2. ***Uncertainty*** – due to unknown sources of emissions
3. ***Methods*** – due to the way the GHG emissions are calculated

The differences between these are discussed in detail in **Chapter 2**, **Section 2.4**, but in summary, "real" variation in LCA results are those that occur due to the inherent variability of a system and "uncertainty" arises due to a lack of knowledge about the true value of a quantity (Björklund, 2002). It is important to distinguish uncertainty from real variability because in theory the former can be reduced by increased measurements or knowledge, whereas the latter is either a natural or an unavoidable characteristic of the modelled system (Ahlgren et al., 2012). Variation due to "methods" refers to the way in which environmental impacts are accounted for in LCA studies, which will depend on the goal and scope of the studies, as it affects the way the analysis is performed and how environmental impacts are attributed to the producer. It can be argued that some of these decisions are 'arbitrary' (Ekvall and Finnveden, 2001). There is a need to understand the importance of such decisions.

The overall GHG balance of a biofuel is particularly important to assess, as they are created in order to directly displace fossil fuels in order to mitigate climate change. Accurately accounting for the net GHG balance of biofuels is therefore a crucial part of determining whether emissions are actually saved.

The aim of this PhD research is to identify the relative importance of LCA methodology in biofuel GHG assessment, and compare this to the impact of variability and uncertainty.

The specific methodology employed in the Renewable Energy Directive (RED), Renewable Transport Fuel Obligation (RTFO) and Publicly Available Standard 2050 (PAS2050) GHG reporting methodologies will be examined in this study as examples of three methodologies that differ in approach. Also, the methods adopted in these methodologies will be critically reviewed to determine whether LCA is being used appropriately in current renewable energy policy.

1.2.1. Outcome and Deliverables

The outcomes and deliverables of this research can be summarised as follows:

- Present a representative case study of 1st and 2nd generation bioethanol production.
- Examine the impact of variability and uncertainty in the GHG results.
- Critically review the main GHG reporting methodologies used in the UK.
- Identify the sensitivities of the results to methodological decisions.

1.2.2. Novel Aspects of this Research

The following are identified as novel contributions from this work:

- This study presents a novel approach to examining LCA methodology in the context of renewable energy and agricultural policy.
- This study provides a novel account of how crop residues are accounted for in GHG reporting methodologies and whether straw removal from soil can compromise the GHG savings potential of straw-based bioethanol.

- A detailed account of the commercial-scale cultivation of Miscanthus is performed using data from growers. The study includes the rhizome multiplication phase.
- This study presents a novel account of the conversion of Miscanthus to bioethanol.
- This study presents a novel theoretical assessment of the GHG implications of harvesting Miscanthus in the autumn or spring.

1.3. Structure of Thesis

Table 1-1 presents the structure of the thesis and the description of each Chapter. The following chapter presents some background and literature review on the production of biofuels and the fundamentals of LCA methodology. The GHG reporting methodologies examined in this research are introduced.

Table 1-1. Outline of the structure of the thesis including a chapter plan.

Stage		Chapter	Description
Research Methodology	2	Background and Literature Review	Provides an overview of the main sources of GHG emissions and LCA methodology.
	3	Research Methodology	Research objectives, description of methods employed
	4	Case studies	Introduction to case studies
	5	Data Collection	Description of data sources used
Model Dataset Development	6	Wheat Cultivation	Results of data collection.
	7	Miscanthus Cultivation	Results of data collection.
	8	Bioethanol Production	Results of data collection.
Results	9	Assessment of Variability and Uncertainty	Assessment of effect of variability and uncertainty on results.
	10	Assessment of GHG Calculation Tools	Comparison of results between tools
	11	Assessment of LCA methodology	Assessment of effect of LCA methodology on results.
Discussion	12	Discussion	Discussion of research results and significance of findings
Conclusion	13	Conclusion	Discussion of research implications, recommendations for further work and improvements

Chapter 2. Background and Literature Review

This chapter provides some background to 1st and 2nd generation bioethanol production, with some reference to the policy drivers that support them and how this has changed over time. The main sources of GHG emissions from bioethanol manufacture are examined, along with a description of the main causes of variation across LCA studies. Details of the methodological aspects of LCA are described in detail. Finally some existing GHG reporting methodologies are introduced and reviewed.

2.1. First Generation Biofuels

Biofuels are similar in physical characteristics to mineral-based transport fuels but differ in that they are produced from plant material (Hammond et al., 2008). They are not a new technology (Nuffield Council on Bioethics, 2011). Historical examples of bioethanol and biodiesel use includes an alcohol-fuelled early automobile in 1895 (Woods & Bauen 2003) and a peanut-oil powered engine in 1890 (Hammond et al., 2008). Interest in biofuels was confined to private projects until in the late 20th century, where it was recognised that such renewable fuels could reduce dependency on, often imported, fossil fuels while stimulating economic development in the agricultural sectors worldwide (Nuffield Council on Bioethics, 2011).

Bioethanol is produced from sugar and starch via fermentation or gasification, though the former is the more commonly pursued route. Biodiesel is produced from the trans-esterification of plant oils (Royal Society, 2008). Biofuels are compatible with most current fuel technology when used in a blend of 5%, although more specialised vehicles can take a higher blend of 85% or even 100% (Hammond et al., 2008). This study focuses on the production of bioethanol, however many of the sustainability issues will be applicable to biodiesel also.

Bioethanol production represents 79% of world-wide biofuel production (EIA, 2011), with the majority being produced in North America. The bioethanol process can be split into three main production pathways, where sucrose (glucose), starch or lignocellulosic material is processed into sugars, which are then fermented (Keshwani, 2009). Utilising readily available sugar or starch-based food crops is relatively straight-forward and this process already takes place in the food and brewing industries. These food-based biofuels are typically referred to as '1st generation' biofuels, as they were the first to be produced at a large scale, and represent 99% of current biofuel production (EIA, 2011).

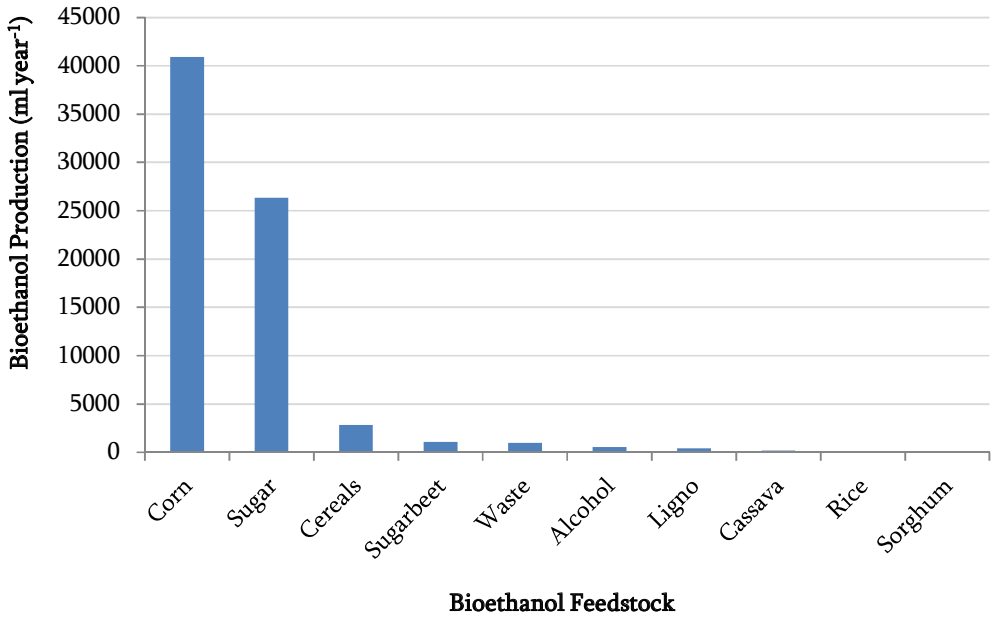

Figure 2-1. The estimated feedstock base of current global bioethanol production (Biofuels Platform 2009).

In current global production, the most popular bioethanol feedstocks are corn in the US, sugarcane in South America and Asia and cereals (mainly wheat) in Europe (Biofuels Platform, 2009, **Figure 2-1**). Sucrose is the easiest feedstock to utilise as the 6-carbon sugars are readily available for fermentation (Keshwani, 2009). The best example of this is ethanol from sugarcane, which is already produced at a large scale (25,000 million litres a year) in Brazil (Biofuels Platform, 2009). In the UK, sugar beet is grown largely for its high sucrose content, and since 2007, British Sugar has used it to produce 70,000 litres of bioethanol per year (British Sugar, 2011).

Corn is the most commonly used starchy-based bioethanol feedstock in the world, 93% of which is utilised in the US. Wheat is an example of this ethanol production pathway in Europe. The UK's current capacity is 895 million litres in 2013, however in 2012 the actual production was only one fifth of this (MacLeay et al., 2013). In 2013 the UK's largest bioethanol plant was opened in Kingston upon Hull by Vivergo Fuels. The plant will produce 420 million litres year^{-1} of fuel at full capacity. The plant will require an annual input of 1.1 million tonnes of feed wheat (Vivergo Fuels, 2013), representing nearly 60% of the total wheat crop from the Yorkshire and Humber area (Martindale, 2009). The plant is expected to produce 500,000 tonnes of animal feed per year, making it the largest single source of animal feed in the UK (Vivergo Fuels, 2013).

2.1.1. Concern Over 1st Generation Biofuels

Despite their promise of a source of low carbon transportation fuel, it could be said that the initial reception to 1st generation biofuels was not positive. In the early 2000's, there were concerns that a major switch to biofuels would lead to competition between the use of crops for food or for fuel; driving up the price of basic food goods, placing a burden on the poor in developing countries who already spend more than half of their income on food (Mitchell 2008). Many of the crops for biofuels were grown in areas with a high density of food-insecure people, who were least likely to require, or benefit from their production (Naylor et al., 2007). A serious concern was that between 2005 and 2008 the price of food commodities more than doubled after the introduction of biofuel production targets (Gilbert 2010). Biofuels were blamed for the food price increase, in both the media and in the literature (Mitchell 2008; Naylor et al. 2007; Rosegrant 2008), though this is a highly controversial claim that is difficult to demonstrate.

After this, food prices continued to rise, appearing to be due to a number of factors: demand for biofuels being one of them. Other causes include China's high demand for soy meal for animal feed, poor harvests due to flooding and drought, a weak dollar exchange rate, and high oil prices (Abbott et al. 2011; Gilbert 2010). Although it is difficult to prove a direct link between biofuels and food prices, two clear trends are evident: total energy consumption will increase in the future, and biofuels will continue to be part of policy targets (Naylor et al., 2007).

There are also concerns that the area of land required to reach biofuel targets will increase the pressure on agricultural land availability, both in the UK and overseas. Increasing pressure on land may have an 'indirect' effect of causing expansion of agricultural land due to an increase in the demand for agricultural commodities, including food, feed and biofuels (Witcover et al., 2013).

The issue of indirect land use change (ILUC) was first identified and highly publicised in 2008 by Fargione et al. (2008) and Searchinger et al. (2008), who suggested that ILUC could have devastating consequences on net GHG emission savings. Searchinger et al. (2008) estimated that if this expansion occurs on areas of high carbon stock, then it could require up to 167 years to 'pay back' the GHG losses from land use change through displacement of fossil fuels. A more extreme payback period of 481 years is estimated for biodiesel produced from soybean on what was previously, forestland (RFA, 2008a). There is of course, no 'payback' when species or habitats are permanently lost. The issue of ILUC has since become prominent in renewable energy policy. In 2009 the first sustainability criteria for biofuels were introduced, stating they could not be grown on areas of high biodiversity or high carbon (EC, 2009a).

2.1.2. Greenhouse Gas Emission Savings from Biofuels

Another major concern with '1st generation' biofuels is that the potential GHG savings were limited by the overall GHG emissions from biofuel production. Originally, in the Energy White Paper of 2003 (DTI et al., 2003), it was stated that biofuels could reduce GHG emissions in the transport sector as they have significantly lower lifetime carbon emissions than fossil fuels.

Previously, in the year 2002 (**Figure 2-2**), the 'Vision for Bioenergy and Bio based Products' in the United States (US) highlighted a primary goal to increase the US's energy supplies by using a more diverse mix of domestic resources that would reduce their dependence on imported oil (BTAC, 2002). Therefore there were conflicting drivers for biofuel development between the UK and US. As a result of this, early life cycle assessment (LCA) studies began to show that the overall GHG balance of biofuels was poor, or even worse than fossil fuels (Marland & Turhollow 1991; Shapouri et al. 2002). This is a result of the biofuel production targets in the US originally being set in place to increase energy security and reduce air pollutants from vehicles, rather than GHG emission mitigation. As a result, corn bioethanol plants were fuelled with average grid electricity, of which, in the US, nearly 50% of which is generated from coal (Wang et al. 2011). As a result, the GHG emissions from bioethanol were worse than conventional gasoline.

Since then, studies have showed that the overall emission savings from '1st generation' biofuels are limited by a number of factors and can vary greatly between producers and supply chains (Menichetti and Otto, 2008). It has been estimated that GHG emission savings, compared to conventional fossil fuels, for wheat-bioethanol range between 10% and 80% (Royal Society, 2008; Woods and Bauen, 2003), 18 and 90% (Quirin et al., 2004), 7 to 77% (E4Tec, 2006) and minus 10% to 38% (Larson, 2006), due to assumptions on management practices, crop yields, crop yield, land-use changes, conversion process efficiencies and LCA methodology. Hence there is large variation in the literature and hence great uncertainty in the potential GHG emission savings from 1st generation biofuels.

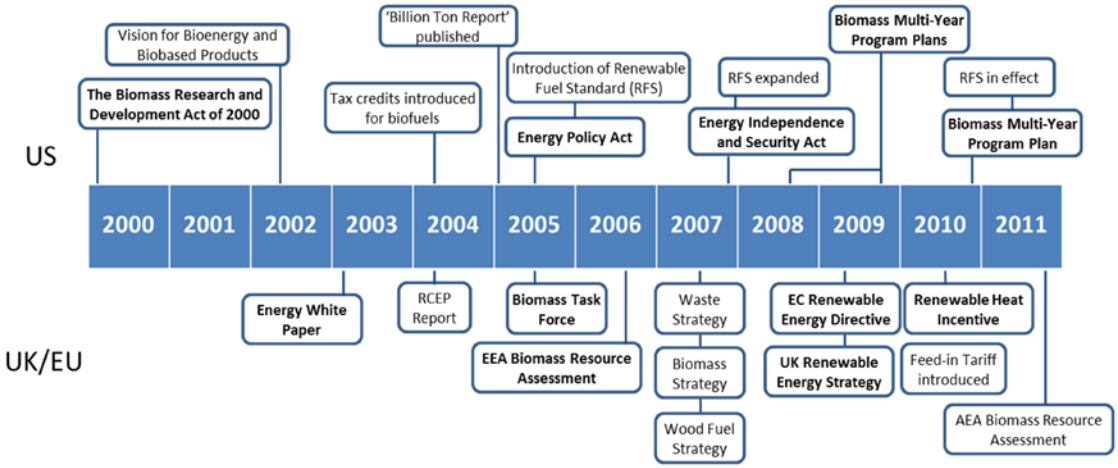

Figure 2-2. Timeline showing biomass policies and strategies in the UK and US since 2000.

2.2. Second Generation Biofuels

Concerns over the potential negative impacts of '1st generation' biofuels on food prices and land use change has led to increased interest of utilising non-food crops as biofuel resources (Singh et al. 2010). Resources such as straw, woody residues from forestry or the waste stream, and purposely grown energy crops can all be used as a feedstock for lignocellulose-based bioethanol. These future biofuels could potentially reduce negative indirect impacts by being more broadly sourced from a range of abundant, readily available and low-cost resources (Akin 2007; Singh et al. 2010). Lignocellulosic bioethanol (and ethanol produced via gasification) is considered to be a '2nd generation' biofuel as it represents a new wave of biofuels, being produced from non-food crops using more recently developed, and still developing, technology (Keshwani, 2009). It is not expected that these biofuels will offer a large-scale alternative to current biofuels before 2020 (Reijnders, 2008a).

Lignocellulosic material is composed of cellulose (35-50%), hemicellulose (20-35%), and lignin (10-25%) which are extensively connected to create a complex, rigid structure that is adapted in order to protect the carbohydrate fraction from decay (Saha, 2004). Hence, lignocellulosic material requires more processing than sucrose or starch-based materials (Akin, 2007). Although there are technical limitations to 5-carbon sugar fermentation, some companies have achieved commercial scale production (Woods and Bauen, 2003) yet there are no fully commercial lignocellulosic plants in operation currently.

Suitable lignocellulosic feedstocks include those with high 5 and 6 carbon sugar compositions (Woods and Bauen, 2003), with a high yield and low cost for production (Hamelinck et al., 2005a). They can be categorised into three main types (Wiloso et al., 2012):

- Residues
- Wastes
- Energy crops

Residues represent a 'non-land' and 'non-food' biomass resource (RFA, 2010), meaning that that they do not directly compete with food production or require land to source them (Wiloso et al., 2012). Residues are an interesting biomass resource as there is an option to remove it from a site, or leave it to decompose. Utilising residues as a biomass resource can open up some key discussion on sustainable use. There is concern that indiscriminate removal of residues can lead to losses of soil organic carbon (SOC) and deplete nutrients from soil (Cherubini, 2010a; Wiloso et al., 2012).

Wastes are assumed to have a zero economic value and are intended to be discarded by the holder of such waste (BSI, 2008a). These also represent 'non-land' and 'non-food' biomass resources. It is assumed that biomass wastes arise are not attributed with the environmental impacts from their original production, and have a zero GHG burden (Wiloso et al., 2012). Diverting material from landfill may also alleviate some of the environmental pressures from methane emissions from anaerobic decomposition of biomass in landfill (Cherubini, 2010a).

Lignocellulosic energy crops are purposely grown for bioenergy purposes, of which short rotation coppice willow and poplar and energy grasses such as Miscanthus are identified as suitable candidates in the UK (DTI et al., 2007). Ideal energy crops are those with efficient solar energy conversion (e.g. C_4 grasses such as *Miscanthus spp.* and switchgrass), low inputs, low water requirements, and a low moisture content at harvest (Cherubini, 2010a).

Originally, it was estimated that 1 million hectares of UK land could be made available for 'non-food' uses, such as arable crops for biofuels and energy crops for heat and power (DTI, 2005). The European Environment Agency estimated that the UK could make 1.2 million ha available for energy crops by growing them on 600,000 ha of set aside land, 148,000ha of sugar beet in 2005 and 6 million ha of grassland (EEA, 2006). These targets were revised to 350,000 hectares by 2020 (DTI et al., 2007). Approximately 93% of land for bioenergy production in Europe comes from arable crops, with only 1% from perennial grasses (EEA, 2013), hence the uptake and use of energy crops is relatively small so far.

Energy crops are usually perennial, being continually harvested over a period of up to 20 years (Bullard and Metcalf, 2001; Elsayed et al., 2003). They are seen as a novel enterprise for UK farmers, both in terms of the methods of cultivation and the role that agriculture has in the energy sector (Sherrington and Moran, 2010). For this reason, the Energy Crop Scheme was introduced in 2000 as part of a Rural Development Programme for England. It aims to encourage farmers to establish energy crops for energy use by providing 50% of the establishment costs (Natural England, 2013a). The success of the scheme is questionable as only 3,356 of the granted 6,376 hectares of Miscanthus were planted between 2000-2006, as well as only 1,815 of the 16,700 hectares of anticipated short rotation coppice (SRC) applications being received (Natural England, n.d.; Sherrington and Moran, 2010). A delay in the update of the scheme meant that virtually no new crops were established in 2006/07, which damaged farmers' confidence in the UK's genuine plans for supporting energy crops (M. Carver pers. com. 2011). Despite this, a further 989 hectares were planned between 2007-2013 (Natural England, 2011), however the scheme is due to end in 2013 (Natural England, 2013b).

A review of literature however shows high variation in GHG emission savings from lignocellulosic bioethanol: such as 15-115% (Quirin et al., 2004), or between -223% and 500% (Borrion et al., 2012a). Reasons for these differences may be difficult to determine, as sometimes the exact feedstock in question is not identified. Specific estimates for wheat straw range between 76-88% (EUCAR et al., 2006), 84% (Elsayed et al., 2003), 13-73% (Borrion et al., 2012b), so variation can still be seen when examining a specific feedstock.

Although such advanced biofuels are not yet in production in the UK, it is important to ensure that these future fuels will be sustainable. Lignocellulosic material does not compete directly for food; however as '2nd generation' biofuels require land to produce them, they are not immune to the potential negative effects of land use change (Cherubini, 2010a).

2.3. Main Sources of GHG Emissions from Biofuels

Due to inconsistencies in the estimated GHG emissions from both 1st and 2nd generation biofuels, a need to assess the GHG balance of biofuels on an individual basis has become apparent (Royal Society, 2008). There have been a number of drivers and policy acts to assess the biomass resources in the UK, and the US (**Figure 2-2**). For biofuels, specifically, the Renewable Energy Directive (RED) in Europe and the Renewable Fuel Standard (RFS) in the US have introduced some sustainability criteria in which to assess the GHG emissions from biofuels (EC, 2009a; EPA, 2010).

Of all economic sectors in the UK, agriculture contributes around 9% of GHG emissions annually (DECC, 2012a), and is a significant component of the lifecycle emissions of many everyday food and other products. Despite there being differences between LCA studies in terms of what crops are used as the feedstock for bioethanol production (e.g. wheat, sugar beet, corn), most studies show that the agricultural and conversion phases account for the majority of the total impacts over the life cycle of biofuels (Menichetti and Otto, 2008).

Despite being a significant component of the life-cycle emissions, there are challenging aspects of quantifying GHG emissions from agriculture, forestry and other land uses (McKone et al., 2011). This is due to the dependence of emission on pedo-climatic and management details which are subject to temporal and spatial variations over various scales; leading to significant uncertainty in GHG emission assessments (Colomb et al., 2012; McKone et al., 2011).

In the literature, the following are identified as major sources of GHG in biofuel supply chains:

- Fertiliser manufacture
- Nitrous oxide emissions from soil
- Land use change
- Fuel for processing and conversion

These are described in the following sub-sections. Irrigation is identified as a major source of GHG emissions (Roches et al., 2010) though this represents just 2% of crops in England (National Statistics, 2011). Diesel fuel usage for farm operations contributes between 5 and 10% of total GHG emissions (Kindred et al. 2008a).

There are also a series of minor sources of GHG emissions:

- Transport GHG emissions are relatively minor in local arable systems (3%, AEA Technology & North Energy Associates 2008). When wheat is imported from abroad the emissions can represent up to 39-56% of total GHG emissions, though this is sensitive to mode, with road transport being the least favourable compared to sea or rail (O'Donnell et al., 2009).
- Pesticide application is shown to represent less than 1% of GHG emissions, based on an assessment of wheat crops in the UK (Webbs et al., 2010). This is because very small quantities are used, and not because the GHG emissions from manufacturing them are

necessarily low (AEA Technology and North Energy Associates, 2010). It is suggested that the use of pesticides in fact improves the GHG balance of arable systems by securing the yield of crops (Berry et al. 2008).

- The use of machinery and equipment typically represents 1% of GHG emissions in arable systems (Bauen et al., 2008), but may be larger (13%) in forestry systems where the machinery used is larger (Whittaker et al., 2011).
- Production of seed is a very minor source of GHG emissions, and consistently represents less than 1% of the GHG emissions in LCA's (Bauen et al., 2008).
- Drying is also a typically a minor source of GHG emissions (Roches et al., 2010).

2.3.1. Fertiliser Manufacture

There are recurrent accounts that nitrogen (N)-based fertilisers cause a significant source of GHG emissions in agricultural systems. This is because both their manufacture and application to soils cause the production of N_2O which has a global warming potential almost 300 times that of CO_2 (Forster et al., 2007). Other phosphorous (P_2O_5) and potassium (K_2O)-based fertilisers are not as significant as they have lower manufacture GHG emissions and they are not associated with N_2O emissions from soils. As assessment of wheat cultivation across the UK showed that nitrogen fertiliser-based GHG emissions accounted for 24% to 62% of total GHG emissions (Webbs et al., 2010). Emissions of N_2O from soil are examined in the following sub-section. This section focuses on fertiliser manufacture, which accounts for approximately 30% of the GHG emissions the cultivation stage (Mortimer et al., 2004).

Nitrogen-based fertilisers include ammonium nitrate and urea, as well as combination fertilisers such as ammonium sulphate, calcium ammonium nitrate and ammonium phosphate (**Figure 2-3**). Ammonia is the precursor to all nitrogen-based fertilisers, therefore is one of the most important synthetic chemical products worldwide (Althaus et al., 2007). It is produced via various modifications of the Haber Bosch process, which is the process by which nitrogen in the air is fixed with hydrogen in the presence of a catalyst. In 85% of cases, the hydrogen is provided from steam-reforming natural gas (Althaus et al., 2007). This is the main contributor of GHG emissions in ammonia production. It is expected that natural gas will continue to be the main feedstock for ammonia production for the next 50 years (EFMA, 2000a).

Ammonium nitrate is the most commonly used nitrogen-based fertiliser in the UK (Thomas, 2011). It is produced by the neutralisation reaction of gaseous ammonium and aqueous nitric acid (EFMA, 2000b). The production of nitric acid is most important step in terms of the overall environmental impacts of fertiliser production (Wood and Cowie, 2004).

There has been increasing pressure in the fertiliser industry to adopt the use of nitrous oxides (NO_X) and N_2O abatement technologies in nitric acid plants. The most commonly used abatement technology is Non-Selective Catalytic Reduction (NSCR), during which natural gas reacts with N_2O to produce CO_2, H_2O and N_2. Such 'best available technology' (BAT) can provide reductions of N_2O by 29% for ammonium nitrate (Brentrup and Palliere, 2008).

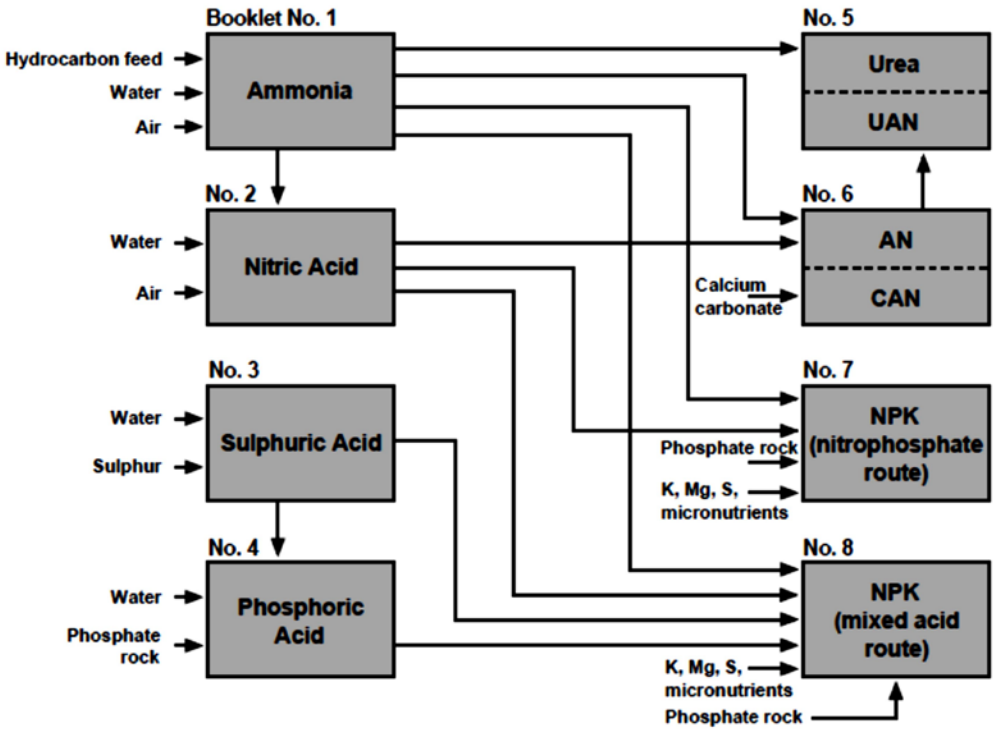

Figure 2-3. The main fertiliser production stages (UAN = Urea ammonium nitrate, AN = ammonium nitrate, CAN = Calcuium ammonium nitrate, NPK = nitrogen, phosphorous and potassium fertiliser) (EFMA, 2000a).

Fertilisers that are P_2O_5 and K_2O-based are less 'GHG intense' to manufacture as they are mined and any processing reactions are not implicated with large emissions of N_2O. Phosphate rock is insolvent and cannot be directly applied to the field, therefore must be applied to farms in form such as mono-calcium phosphate, superphosphate or triple super phosphate. The production of these fertilisers involves the reaction of phosphate rock with either sulphuric or phosphoric acid (EFMA, 2000c). The most commonly found potassium ore is potassium chloride, or potassium salts, or potash. After concentration, cleaning and crushing it can be applied either directly to soil, or in a combination NPK fertiliser (EFMA, 2000c).

2.3.2. Emissions of N_2O from Soil

It is estimated that N_2O emissions account for 8% of total global greenhouse gas emissions (Bernstein et al., 2007), of which 61% of are estimated to arise directly and indirectly from the agricultural sectors, mainly from N fertiliser use (European Commission et al., 2010). In the UK, N_2O emissions are known to contribute 38% of the total GHG emissions from the agricultural sector (HM Treasury, 2006). Nitrous oxide is a natural product of the nitrogen cycle (**Figure 2-4**), however it can be stimulated by application of nitrogen to soil (De Klein et al., 2006). The rate at which is released is highly uncertain (Kindred et al. 2008a; Rowe et al. 2011; Whitaker et al. 2010; Yan & Boies 2013).

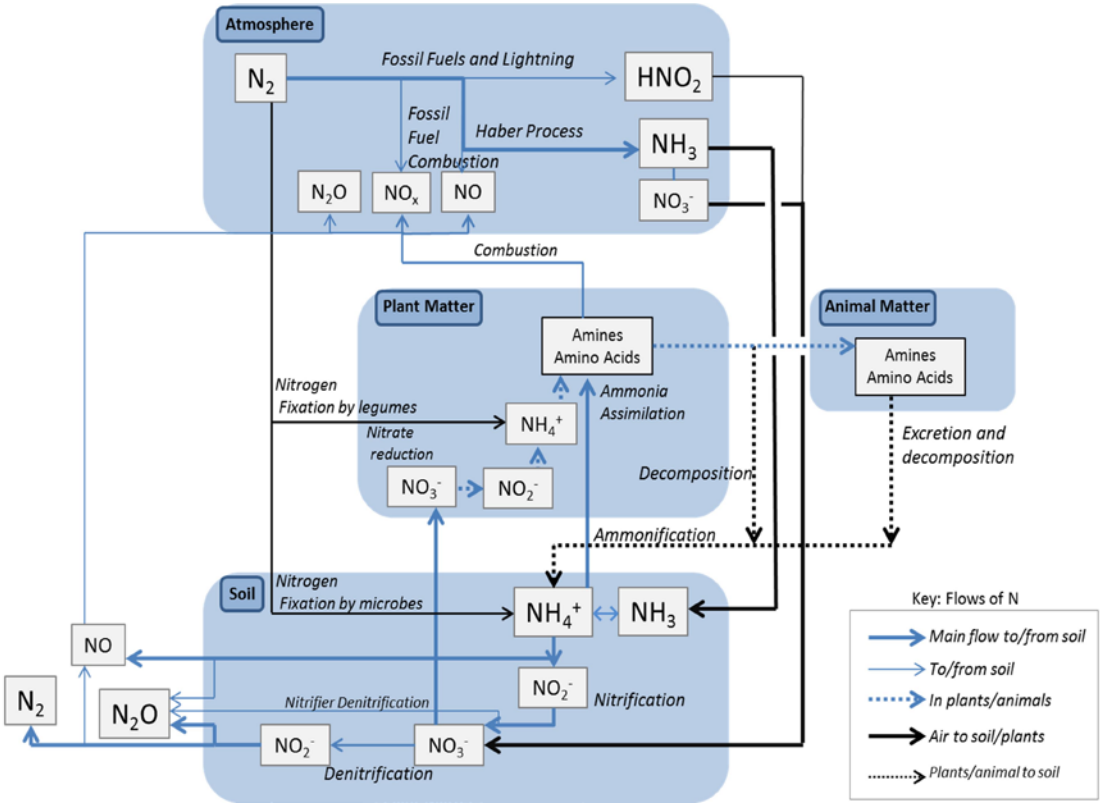

Figure 2-4. Overview of the nitrogen cycle.

Nitrogen is essential for plant growth, as it is required for synthesis of basic building blocks for cells including amino acid and proteins (Huo et al., 2011). The nitrogen cycle has been described as being one of the most controversial nutrient cycles to decipher (Clark, 1979). It describes the many forms that nitrogen can exist in the environment, whether in the air, plant material or soil, and how it passes between each stage through the processes of growth, decay, fixation, nitrification and denitrification (**Figure 2-4**). Interestingly, nitrogen is found in chemical forms that are, to biological organisms, either readily utilisable, or completely benign. For plants to be able to utilise it, it must be converted to nitrate ions (NO_3^-). Once absorbed, the nitrates undergo reduction to ammonium (NH_4^+), where it can used for plant development and growth (Delwiche, 1983).

Nitrogen fixation occurs when dinitrogen (N_2) is converted to ammonia either via the Haber Bosch Process or through nitrogen fixation by microorganisms. Another route of nitrogen fixation is via ionisation of N_2 and O_2 to NO^+ during thunderstorms, producing nitrous acid (HNO_2) which can form nitrates in the soil (Ferguson & Libby 1971; Li et al. 2002), though the contribution is negligible (Cooray et al., 2009). Fossil fuel combustion also fixes N_2 from the atmosphere at high temperatures in combustion engines (Vitousek et al., 1997). Fixation in engines is estimated to contribute almost 9% to total anthropogenic N_2O emissions in 2008 (European Commission et al., 2010).

Ammonification is the process whereby organic nitrogen integrated within dead plants, animals and manure is converted to inorganic ammonium by soil microorganisms during the decomposition process, therefore is a key process of nutrient recycling in ecosystems (Eghball et al., 2002). The rate of mineralisation is affected by the quality of plant material, soil temperature and moisture, and abundance of decomposers in the soil (Dresbøll and Thorup-Kristensen, 2005).

Nitrification is carried out by organisms that derive energy from the oxidation of ammonia, and nitrite. The process effectively mobilises N in the soil, as the ammonium cation ($NH4^+$) has a greater tendency to bond with organic soil matter (Delwiche, 1983). Nitrous oxide is formed during oxidation of both ammonia and nitrites (**Figure 2-4**), though the rate at which this occurs is not well understood, and difficult to examine in intact soils.

Denitrification is the process whereby soil microorganisms utilise nitrates as a substitute for oxygen. It causes the production of N_2O, N_2 and NO and causes, in effect, a major loss of nitrates from soil, decreasing the efficiency of fertiliser and animal manure application (Knowles, 1982). The process can be inhibited in low pH and aerobic conditions (Wrage et al., 2001). There is evidence that soil temperature and high organic C content has a positive effect on denitrification (Bremner & Shaw 1958; De Klein & Van Logtestijn 1996; Miller et al. 2008).

Historically, the process of examining the nitrogen cycle has involved first deducing the main processes involved, estimating budgeted flows of nitrogen, and then developing simulation models to predict flows between stages (Clark, 1979).

Significant contributions to this field of knowledge were made by Alexander Felix Bouwman. In 1996, Bouwman discovered significant correlations between N fertilisation and N_2O emissions, including impacts from the time period, crop and fertilizer types and soil properties. From a sample of 263 fields from 44 references, the fertiliser induced emission (FIE) averaged 0.6%, ranging between 0% and 7%.

By using least fitting squares to plot a linear relationship **Figure 2-5**, **Equation 2-1** was deduced, which shows high correlation (r^2 from linear regression is 0.8). At a given location, the soil water content, N content and soil temperature are most likely to be the main factors affecting N cycling; however between different locations pH and available soil carbon also become important (De Klein and Van Logtestijn, 1996).

Equation 2-1. Estimating N_2O-N emissions (E) from soil based on the relationship in Figure 2-5.

$$E = 1 + 0.0125 * F$$

This is known as a 'direct emission' of N_2O. Where E is the total annual emissions of kg N_2O-N, which is the N component of N_2O, and F is applied N fertiliser (kg N ha^{-1} $year^{-1}$).

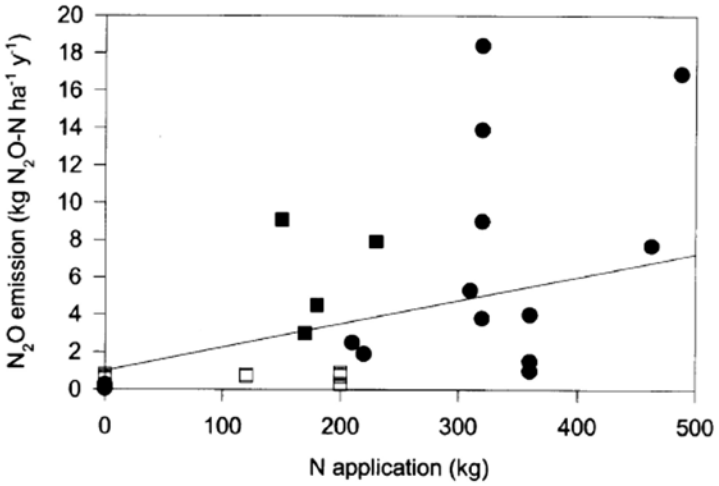

Figure 2-5. The annual N_2O-N emissions from specific N fertiliser application. Continuous line is regression line from Bouwman's (1996) analysis of emission factors ● grassland; ■ arable (non-cereal) crops; □ cereal crops (Smith et al. 2003).

The value of 1 represents the background N deposition rate (1 kg N ha^{-1} $year^{-1}$). Based on Bouwman's dataset, the FIE ranges between 0.25 to 2.25%. The uncertainty range was applied in the IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (Smith et al. 2003), however was revised to 0.3 to 3% in the updated IPCC guidelines for National Greenhouse Gas Inventories (De Klein et al., 2006), in the light of updated research by Bouwman et al. (2002).

Similar models have been developed to examine N_2O emissions that occur due to volatisation and leaching of nitrogen from soils, which are known as indirect N_2O emissions. To convert N_2O-N to N_2O, it must be multiplied by the molecular weight ratio of between the two (**Equation 2-2**).

Equation 2-2. Converting N_2O-N to N_2O (De Klein et al., 2006).

$$kg\ N_2O = kg\ N_2O\text{-}N * [\ ((2*14) + 16) / (2*14)\ ]$$

$$kg\ N_2O = kg\ N_2O\text{-}N * (44/28)$$

In summary, performing accurate measurements of N_2O emissions requires intensive experimentation; hence it is advantageous to develop simpler methods to predict soil processes under different conditions (De Klein and Van Logtestijn, 1996). The defaults in the IPCC calculation guidelines for national reporting tend to 'average out' much of the climatic and geographic variation that N_2O emissions from soil are sensitive to, and tends to provide a high-level estimate (Hillier et al., 2011).

2.3.3. Land Use Change

Land use change (LUC) is believed to cause 15% of global GHG emissions (Witcover et al., 2013). It occurs when a given use or management of land is changed (Bickel et al., 2006), and can occur either directly (DLUC), or indirectly (ILUC) as a consequence of changes in production rates and prices of other commodities (Sanchez et al. 2012,). Soil organic carbon (SOC) sequestration occurs as a result of the long-term storage of atmospheric CO_2 as a relatively inert form of carbon with a potential residence time of decades to centuries (Kochsiek and Knops, 2012; Lal, 2008a). Losses can occur due to oxidation of SOC after tillage of land, as well as through the burning or removal of standing and dead biomass (Lal, 2004a).

A general feature of ILUC is that their impacts are more complex and uncertain than that from the production of biofuels (Overmars et al., 2011). Although ILUC is, ultimately, the same as DLUC, the main difference is the uncertainty of where this ILUC event occurs, or the amount of land that is required to compensate for losses of arable land to bioenergy. For example, a Miscanthus crop can be grown on what was previously agricultural land (**Figure 2-6**). This is DLUC, however ILUC may occur now that some agricultural land can no longer be used for food or feed production. It is assumed another area of agricultural land must be created in order to maintain a constant supply of food or feed. Due to global trade of agricultural commodities, ILUC can occur anywhere in the world. Therefore, it is possible for a biofuel crop to cause both DLUC and ILUC. The following sections describe DLUC and ILUC.

2.3.3.1. Direct Land Use Change (DLUC)

In current GHG reporting and renewable energy policy, the main focus has been on DLUC (Overmars et al., 2011). The RED requires that any carbon stock change is accounted for in GHG reporting of biofuels (EC, 2009a). Calculating this requires a great deal of data collection on the volumes of above and below ground biomass before and after conversion, carbon storage in the biomass and the soil and the effect of local management and conditions before and after conversion (Lange, 2011). More simplified methods are presented in a later supporting document (EC, 2010a), which has been examined in detail in Lange (2011). Other literature resources (Hillier et al., 2011; St Clair et al., 2008) have provided accounts of specific LUC events, and the Cool Farm Tool contains in-built calculations for LUC (Hillier et al., 2011).

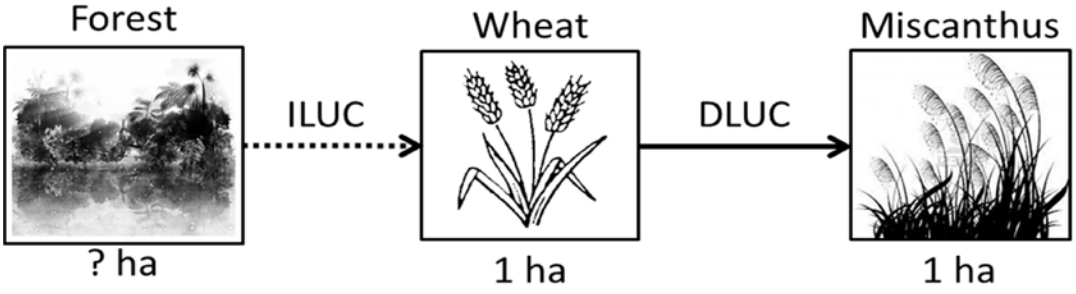

Figure 2-6. DLUC occurs when one land use is directly changed, yet this can cause indirect changes and with unknown consequences (ILUC).

The emissions from DLUC are temporarily allocated between a period of 20 years to comply with specifications in the IPCC Guidelines (**Figure 2-7**). For example, forestland, grassland and arable land have progressively lower equilibrium SOC, therefore the SOC losses are incurred if forestland or grassland are converted to arable land (St Clair et al., 2008). Conversion of natural land for bioenergy production occurs has shown to compromise the ability for bioenergy to reach 35% GHG emission reduction targets set in the RED (Lange, 2011). There is evidence that energy crops grown on arable land can sequester SOC (Hillier et al., 2009).

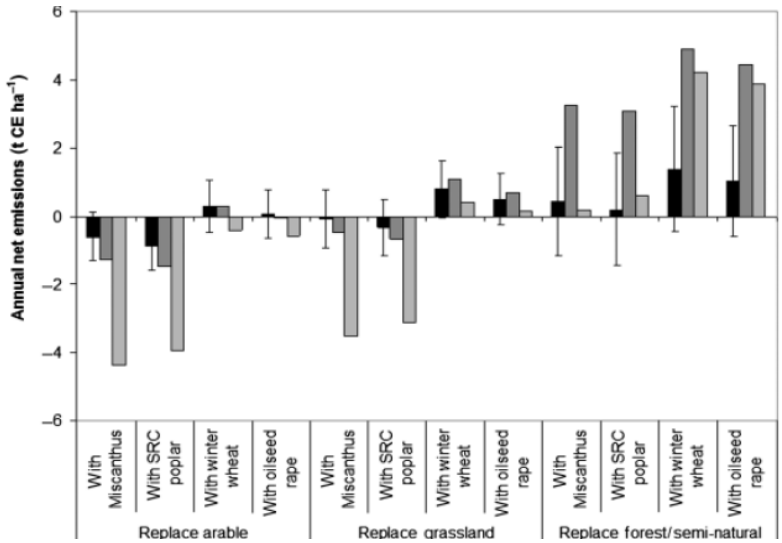

Figure 2-7. The net GHG emissions for replacement scenarios (Hillier et al., 2009).

Changes in residue management are less clear-cut compared to LUC events (Powlson et al. 2011). There is on-going debate to the impacts of straw removal from soil (Cherubini and Ulgiati, 2010), and few studies have placed these impacts in the context of a LCA study. The majority of studies examine impacts on soil quality; there is currently no consensus (Cherubini and Ulgiati, 2010; Gabrielle and Gagnaire, 2008). One study examined the impact of straw removal in the context of bioethanol production from wheat straw and found that net GHG emissions savings of 49% could be achieved (Cherubini and Ulgiati, 2010). This was due to a combination of increased fertiliser requirements to replace lost nutrients and due to foregone SOC land use change due to straw removal. Their calculations show that straw-based bioethanol could fulfil the 35% GHG emission saving target set by the RED (for installations established before January 2017), but not the 60% target that must be met after January 2018. Therefore the GHG implications of straw removal could contribute at least 50% of the emissions of bioethanol production. Their results suggests that the GHG implications of straw removal may compromise the ability for straw-based bioethanol to achieve future RED emission saving targets. **This is identified as a gap in current knowledge and will be explored in this study.**

2.3.3.2. Indirect Land Use Change (ILUC)

The issue of ILUC was first identified and highly publicised in 2008 by (Fargione et al., 2008; Searchinger et al., 2008), who suggested that ILUC effects of using land for biofuels could have devastating consequences on net GHG emission savings. Searchinger et al. (2008) estimated that if ILUC stimulates the conversion of areas of high carbon stock, then it could require up to 167 years to 'pay back' the GHG losses through displacement of fossil fuels.

It is suggested that the effects of ILUC on the GHG emission from biofuel supply chains can be avoided or reduced by promoting the use of feedstocks that require less land (wastes and residues), agricultural intensification, yield increases, or making use of co-products from biofuel production for animal feed (Dehue et al., 2009; Witcover et al., 2013). It is also possible to increase the agricultural land base by converting unproductive land (Overmars et al., 2011), though there may be yield limitations in such areas (Lesschen et al., 2012). It is suggested that future 2nd generation biofuel produced from lignocellulosic feedstocks could reduce negative indirect impacts of biofuels by being more broadly sourced from a range of abundant, readily available and low-cost resources that do not directly compete with food (Akin 2007; Singh et al. 2010).

Calculating ILUC is a more complex process than for DLUC (Sanchez et al. 2012; Witcover et al. 2013), mainly because it is impossible to monitor and attribute to a specific cause (Overmars et al., 2011). For this reason, it is not included in current GHG reporting methodologies for biofuel regulation. Modelling ILUC requires a coupled modelling framework that estimates interaction of agricultural and energy markets, predicts production volumes and prices, and estimation of risk of conversion of land (Bauen et al., 2010; Witcover et al., 2013). The GHG implications of ILUC will also depend on the types of land conversions occurring (Searchinger et al., 2008). Models can either be based on economic models (Hiederer et al., 2010), patterns of land use change (Bauen et al., 2010) or statistics (Kim & Dale 2011).

2.3.4. Biofuel Conversion

Electricity and heat requirements for processing, drying, and conversion can contribute between 17% and 28% of total GHG emissions per MJ biofuel (Punter et al., 2004). The quantity and type of process energy can significantly affect the overall results (Menichetti and Otto, 2008). For example, if coal is used to provide heat and power the GHG emissions are 3% higher than conventional gasoline (Wang et al. 2012). Emissions from conversion can be reduced by utilising a renewable energy source for heat and power (Elsayed et al., 2003; Punter et al., 2004), such as biomass, whose combustion of biogenic carbon (that originating from biomass combustion) is considered to be carbon neutral. A combined heat and power (CHP) plant can reduce the overall energy requirements by 15% to the 'outdated practice' of utilising separate natural gas boilers and taking electricity from the grid (EUCAR et al., 2006). In this case, the CHP plant is scaled in order to satisfy the heat demands of the process, and there can be a surplus of electricity generated that can, in theory, displace some grid electricity (Punter et al., 2004).

2.4. Causes of Variation in LCA's of Biofuels

Chapter 1 introduced the following major causes of variation of LCA results:

- ***Real variation*** – due to real differences in supply chains
- ***Uncertainty*** – due to unknown sources of emissions
- ***Methods*** – due to the way the GHG emissions are calculated

Understanding the relative importance of each of these sources of variation will aid the process of reducing variability between studies. **This is identified as a gap in current knowledge and will be explored in this study.** Real variation and uncertainty can, in theory, be reduced through increased measurements and knowledge, whereas reducing variation due to methodology should be relatively straight forward through standardisation of methods.

These causes of variation are introduced in the following sections.

2.4.1. Real Variation

"Real" variation in LCA results are those that occur due to the inherent variability of a system (Björklund, 2002). The GHG emissions of a biofuel supply chain will depend on a multitude of factors that contribute to the final result. There may be variation in agronomic practices, such as site inputs, cultivation intensities and yields, different fuel requirements for processing or drying, and varying transport distances with different scales. Understanding this source of variation may help highlight where policies could target the action of GHG mitigation strategies (Rowe et al., 2011).

Real variation can exist both spatially and temporally (Björklund, 2002). Spatial variation can be examined at various scales, such as between different locations on a field, in a country, or even across countries. An example of a spatial variation-based study was published by Webbs et al. (2010), who assessed farming practices across the UK to determine the relative GHG emission savings from bioethanol. Temporal variation can exist at various scales, as LCA studies may consider periods of time from one year to many, for example crop rotations. Studies can also back-calculate or predict future production scenarios to consider trends of emissions over time.

It has been demonstrated that LCA results are highly sensitive to variation in yields, nitrogen fertiliser use and the process energy source to the biofuel conversion process (Ahlgren et al., 2012; Menichetti and Otto, 2008; Whitaker et al., 2010). Higher yields mean the environmental impacts are shared between more products (Röös et al., 2010). Electricity and heat requirements for processing, drying, and conversion can contribute a large proportion of the emissions as they may or may not involve the combustion of fossil fuels. There is, however, a great deal of uncertainty associated with some of these impacts, and this is also a cause of variation in LCA studies, as discussed in the following sub-section.

2.4.2. Variation Due to Uncertainty

In a LCA study, "uncertainty" arises due to a lack of knowledge about the true value of a quantity (Björklund, 2002). It is important to distinguish uncertainty from real variability because in theory the former can be reduced by increased measurements or knowledge, whereas the latter is either a natural or an unavoidable characteristic of the modelled system (Ahlgren et al., 2012). Uncertainty can appear in many forms, including:

1. ***Data inaccuracy or inadequacy*** – where there is either error in the collected data meaning it is not representative to the system studied. This can be because there are gaps in the data, it is collected incorrectly, or from a related process or from another country.
2. ***Model uncertainty*** – sometimes, aggregation of spatially or temporally ranging datasets can lead to oversimplification of the data.
3. ***Uncertainty due to choices*** – concerns decisions that are made when the LCA studies are modelled. This particularly concerns methodological variation, as there can be uncertainty in the decisions made on co-product allocation and functional units.
4. ***Epistemological uncertainty*** – this is caused by lack of knowledge about a system and can be reduced by measurements and research.

Processes that take place in the soil are considered to be the main source of epistemological uncertainty in biofuel supply chains, particularly regarding N_2O emission rates from soils (Whitaker et al., 2010). These have been demonstrated to cause a significant amount of both variation and uncertainty in the GHG emission results in LCA studies of biofuels (Menichetti & Otto 2008; Kindred et al. 2008a; Rowe et al. 2011; Yan & Boies 2013), as well as other arable crops (Ahlgren et al., 2012; Guo et al., 2011; Röös et al., 2010).

It is possible that epistemological uncertainty can be reduced by applying average, or default numbers. For example, the Intergovernmental Panel on Climate Change (IPCC) provides some default emission rates of N_2O emissions from soils (De Klein et al., 2006). They classify the defaults into three Tiers. Tier 1 emission factors use average default values that are derived to be applicable at global or national scale. Tier 2 methods increase the level of detail by employing "smart" emission factors that are specific to particular technologies or regions. Tier 3 methods incorporate increasingly more complicated or involved methods such as process-based models or direct measurement, for example DAYCENT (Del Grosso et al. 2001) or DeNitrification-DeComposition (DNDC) model (Li et al. 2011). Using such tools requires a greater understanding of soil and plant systems compared to using the national default values (Hillier et al., 2011).

A study by Guo et al. (2011) found that the IPCC Tier 1 approach gave a much higher GHG emission result compared to more detailed modelled results using the DNDC model. Another study discovered a 1.7-fold overestimation by IPCC Tier 1 compared to DNDC (Brown et al. 2002). Likewise, a similar study by Yan & Boies (2013) showed the DNDC reduced the overall level of uncertainty by 18-26%.

2.4.3. Variation Due to LCA Methodology

Uncertainty due to choices was highlighted as a potential source of uncertainty in LCA studies (Björklund, 2002). Likewise, it is acknowledged that inconsistencies with methodological approaches can cause variation between studies and makes it difficult to compare them (Reijnders, 2008a; von Blottnitz and Curran, 2007). Methodologies can determine how environmental impacts are accounted for in LCA studies. Before methodological differences in LCA's can be examined one must first define the framework and approach of a LCA study.

2.5. Life Cycle Assessment Framework and Approach

LCA is an integrated and iterative process (**Figure 2-8**) that systematically accounts for all the inputs that arise from the production, use and disposal of a product (CEN, 2006a). The ISO standards 14040:2006 (CEN, 2006a) and ISO 14044:2006 (CEN, 2006b), describe the main phases of performing a LCA as determining the goal and scope, performing an inventory analysis, impact assessment and results interpretation (shown in **Figure 2-8**).

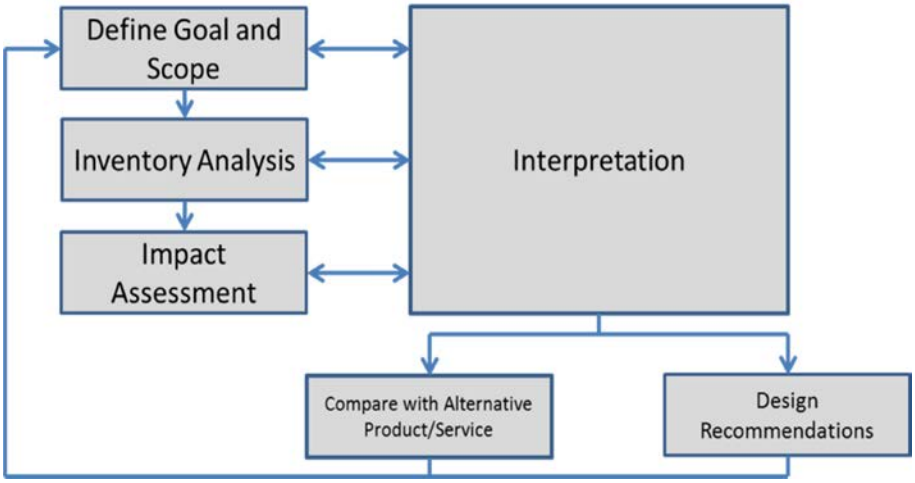

Figure 2-8. A process flow diagram of LCA (Kelly et al. 2012).

In summary, the process of defining the **goal and scope** of a LCA study involves outlining the main aim of the study and considers the stages and sources of environmental impacts included in the analysis (CEN, 2006a). The **inventory analysis** involves quantifying the inputs and outputs of materials in the system studied. The purpose of **life cycle impact assessment (LCIA)** is to translate the inventory data into an environmental burden. The LCIA involves first identifying the emissions that arise from the studied system and classifying them into environmental impact categories. The emissions must then be **characterised** to represent a single unit of measurement whereby to assess the impact. Both classification and characterisation are minimum requirements in LCA's (CEN, 2006a). Additional options include normalisation and weighing, but these are not carried out here.

2.5.1. Goal and Scope Definition: Attributional and Consequential LCA

As the goal and scope is the first step in performing a LCA, it should influence the forthcoming phases (Tillman, 2000). All proceeding phases will be shaped by decisions made in the goal and scope definition. For example, the level and quality of data collected must be sufficient in order to fulfil the original aim of the study (Singh et al. 2010). The aim is usually defined into a functional unit, which defines the quantity and function of the finished product for analysis. The impact assessment must reflect the environmental impact categories defined in the goal and scope. The interpretation of the results must also reflect the original goal and scope.

LCA's tend to map or account for the emissions that a product or service is accountable for (Sandén and Karlström, 2007), however this can be examined in two main ways:

- **Attributional LCA** - provides information about the direct GHG emissions that are directly attributed to the production and use of a product (Sanchez et al., 2012).
- **Consequential LCA** - examines the GHG emissions that occur due to a change in production of a product (Brander et al. 2009a).

The difference between these two approaches can be demonstrated by a study performed by the Nuffield Council on Bioethics (2011), which structure the following questions with regard to the environmental assessment of biofuel supply chains:

- **ALCA** – "Who is responsible for any given net change in total GHG emissions due to biofuels production?"
- **CLCA** – "What is the overall effect on GHG emissions of a policy which promotes the production of biofuels?"

This highlights that there is an element of responsibility to consider when establishing a method in which to regulate GHG emissions from biofuels. It is recommended that biofuels are certified according to an ALCA approach, because it calculates total direct emissions from a product (Sanchez et al., 2012), and producers have immediate control over any direct emissions they cause during production (Nuffield Council on Bioethics, 2011). An ALCA approach is also useful for identifying opportunities for reducing emission within the life cycle or supply chain (Brander et al. 2009a).

A CLCA approach models emissions due to changes in outputs of a product (Sanchez et al., 2012), so that the overall impact of implanting the biofuel targets is considered in a wider, even global context of producers and consumers (Nuffield Council on Bioethics, 2011). A CLCA approach is considered to be the appropriate method for policy makers (Brander et al. 2009a).

In summary, the goal and scope of a given LCA will depend on whether it takes an attributional or consequential approach, and it is reasonable that they will have different LCA methodological approaches (Aylott et al., 2011). The main stages of a LCA are described in the following sub-sections with the differences between ALCA and CLCA in mind.

2.5.2. The System Boundaries

The system boundaries of a LCA study are defined according to the goal and scope of the study (Singh et al. 2010). The boundaries of the study are very important as differences can cause discrepancy between studies (Bird et al., 2011). The boundaries should clearly define the stages of a product's lifecycle and sources of environmental impacts that are accounted for. It is recommended that some cut-off criteria are used when determining the system boundaries of a study, as it is rarely possible to capture information on the entire system (CEN, 2006b). There are many minor sources of GHG emissions which add a very small contribution to the overall footprint. Care must be taken however, as there is evidence that in many biofuel chains, the smaller contributors can add up to contribute between 10-20% of the overall footprint (Bauen et al., 2008). The ISO standards recommend the following cut-off methods in the system boundaries:

- **Mass**- the given contribution of a material to the mass flow of the product should be included in the study if it surpasses a pre-determined cut-off value.
- **Energy** – a process should be included in the study if it contributes more than a pre-determined cut-off value in terms of energy requirements.
- **Environmental significance** – a process should be included if it is implicated with more than a defined amount of the environmental impact of a product.

Determining whether a given stage or output of emissions falls above or below a pre-determined cut-off criterion may require some prior knowledge of the system. This could be determined by performing either a literature review or a "hot-spot" analysis to determine the main causes of GHG emissions in a process. It is suggested that sources that consistently contribute less than 1% in other studies should be excluded. In biomass supply chains such emissions include production of seed, manufacture and maintenance of machinery and equipment, emissions of perfluorocarbons, hydrofluorocarbons, sulphur hexafluoride and chemicals used in small quantities (Bauen et al., 2008). It is suggested in the PAS2050 methodology that an LCA should account for at least 95% of the 'anticipated life GHG emissions' (BSI, 2008b).

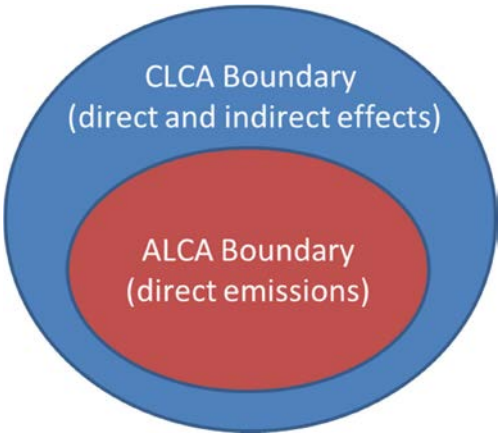

Figure 2-9. The system boundaries for ALCA and CLCA (Brander et al. 2009a).

There are also differences in the system boundaries between studies depending on whether an ALCA or CLCA approach is adopted (**Figure 2-9**).

- An ALCA study would include all system flows under direct or indirect control of the operator. It would not include co-product processing in the system boundaries (Aylott et al., 2012). Allocation should occur at the point of production (Kindred et al., 2008a).
- A CLCA would examine flows within and outside of the product's system boundaries to examine indirect effects such as those on the market, including substitution and resource constraints (Brander, et al. 2009a; Sanchez et al. 2012).

2.5.3. Inventory Analysis

The inventory analysis phase represents the process of collecting data. The stage can be the most time consuming, and hence expensive, phase of performing an LCA (Carbon Trust, 2008). The main issues surrounding inventory analysis include data collection, estimations, validation of data and relating data to specific system boundaries (Singh et al. 2010). Production systems rarely involve single producers, so data may need to be collected from a number of partners. Requirements for data quality will also place demands on the data collection phase, when aggregated or estimated data is all that can be accessed. Therefore, this process can be challenging.

The following stages are performed when carrying out the inventory data collection:

1. **Scoping the supply chain** – the supply chain should be defined according to the system boundaries of the study, which are determined by the goal and scope. This should include all phases between the defined 'cradle' and 'grave' of a product. For example, in biofuel production, this would include the cultivation of the crop to the combustion in a vehicle.
2. **Identifying the mass flow** – involves quantifying the inputs and outputs to each phase of the supply chain. This should include an account of the condition, or quality of the material at each point, such as moisture content or value.
3. **Identifying emission factors** – emission factors are multipliers which translate units of materials into GHG emissions. This is part of the impact assessment phase (discussed in the following section).

There are some differences between an ALCA and CLCA in the data inventory phase, apart from those caused by different system boundaries. It is considered appropriate to use values that represent average production when performing an ALCA, or alternatively default numbers can be used (Brander, et al. 2009a; Sanchez et al. 2012). A CLCA will tend to use marginal data, but will also require models on price elasticities, product demand and supply curves as well data used in the ALCA (Sanchez et al., 2012).

2.5.4. Impact Assessment

The impact assessment phase establishes a relationship between the product and its impacts on the environment (Singh et al. 2010). This includes the consumption of primary energy and raw material. It also includes emissions that occur due to combustion of fuels, chemical reactions from processes and reactions and biological processes.

In LCA, there are numerous environmental impacts that can be monitored (**Figure 2-10**). The impact assessment phase also involves characterisation, whereby the environmental impacts are categorised according to their corresponding environmental impact category and attributed with a relative weighting (CEN, 2006a). The totalled result for that environmental category is known as the 'Midpoint' result (Goedkoop et al., 2009). For example, greenhouse gases are placed into the category which is implicated with climate change. Within this category, the greenhouse gases are weighted according to their relative global warming potential. Typically, this is characterised according the 2007 IPCC global warming potentials (Forster et al., 2007) which are as following:

- Carbon dioxide (CO_2) : 1
- Methane (CH_4) : 25
- Nitrous oxide (N_2O): 298

The total quantity of CO_2 equivalents (eq.) released due to the production of a product is the midpoint result for the environmental category 'climate change'. These results can then be aggregated to higher-level 'Endpoint' impacts, which apply environmental models to estimate the larger potential implications that the product studied may cause. For example, in the Recipe Methodology by Goedkoop et al. (2009), the midpoint results can be used to predict damage to human health, ecosystems and resource availability (**Figure 2-10**). In general, there is more uncertainty associated with endpoint analysis, due to the uncertainty involved in modelling actual physical impacts that emissions from one process may cause. It is suggested that endpoint analysis provides a mechanism to communicate the results from an LCA study to non-practitioners, whereas a midpoint assessment generates a single-number answer which may provide less information (Goedkoop et al., 2009).

In the case of current renewable energy policy, the main driver has been for regulation of emissions implicated with climate change, hence these are studied here. The study is therefore 'selective' as only select environmental impacts are examined. Here a midpoint analysis is sufficient because the GHG emissions from biofuels are being directly compared to those of gasoline.

Impact assessment often relies on the use of emission factors. These are based on LCA studies in themselves, of which the results are stored in databases, such as Ecoinvent, which is a peer-reviewed database (EcoInvent, 2007). These databases can provide a short-cut to estimating the GHG emissions from a process, although they may not always be representative to the system studied.

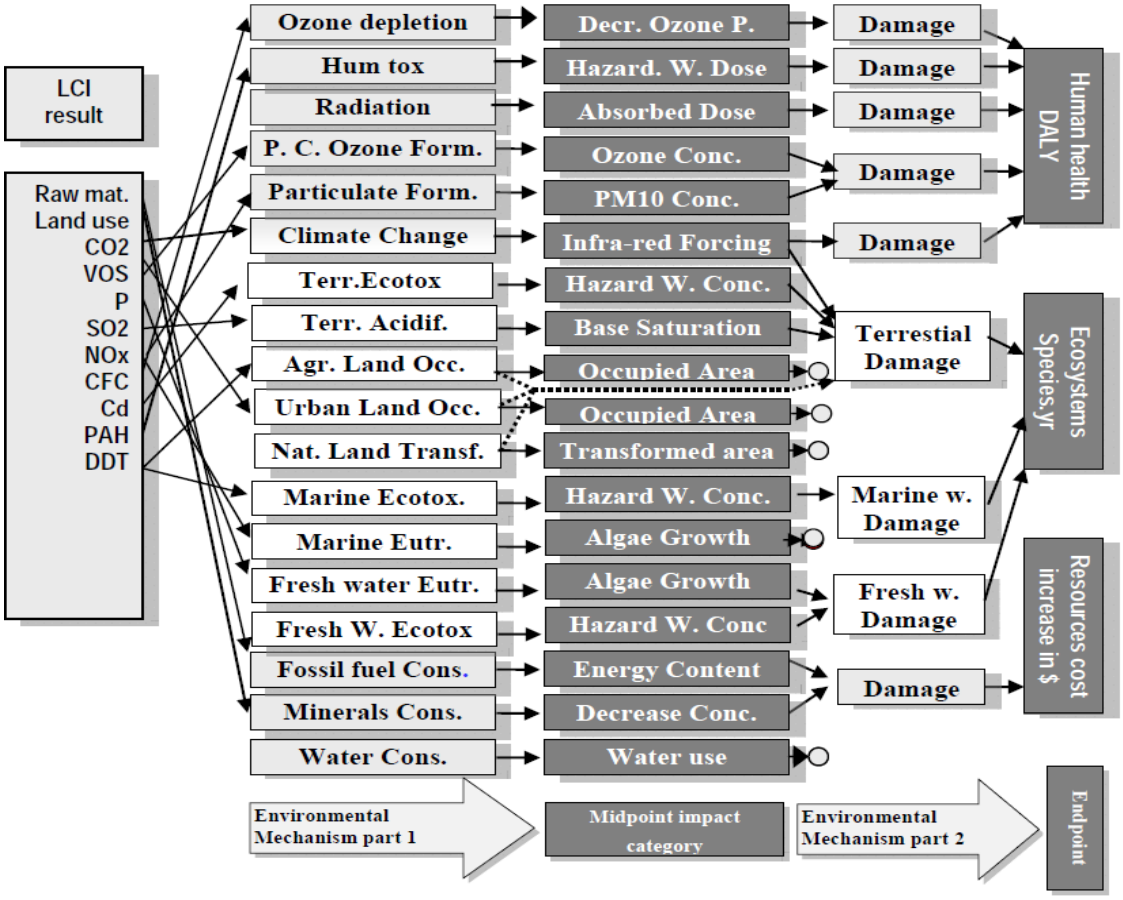

Figure 2-10. The environmental impacts and the relations between the LCA parameter (left), midpoint indicator (middle) and endpoint (right) (Goedkoop et al. 2009).

2.5.5. Dealing with Co-Products in LCA

Before it can be decided how to treat co-products in LCA calculations, one must first determine whether a material is indeed a co-product or a waste (Weidema 2003).

2.5.5.1. Identifying Co-Products and Wastes

Distinguishing between wastes and co-products is crucial in LCA. The treatment of waste is the responsibly of the main product. A co-product would be allocated emissions from the upstream production stream, alternatively system expansion is performed (**Figure 2-11**).

When examining the existing legislative definitions of waste, it is reiterated that a waste is "any substance or object which the holder discards or intends or is required to discard" (BSI, 2008a; CEN, 1998; EC, 1996). There is no clear guidance on when a material becomes, and ceases to be, a waste (Brander et al. 2009b). A material can be a true waste but still be 'capable of economic reutilisation' (Brander et al. 2009b), suggesting that a substance or object could still be classed as a waste despite it having an economic value or not.

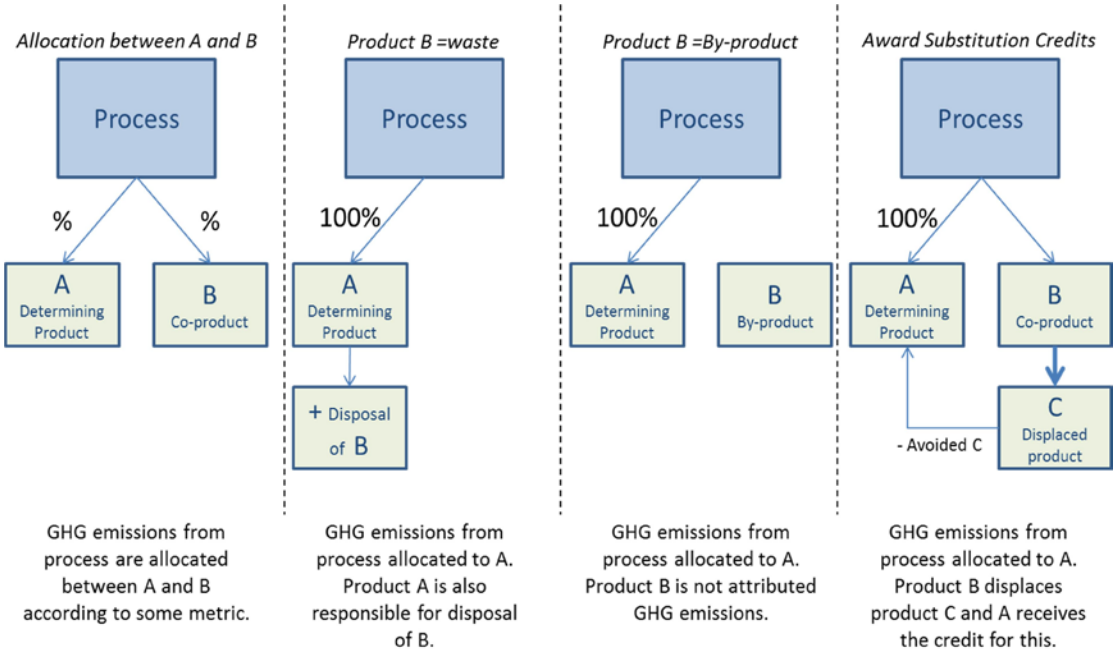

Figure 2-11. Demonstration to how co-products are treated in LCA methodology, using an example process yielding two products, A and B.

Residues produced from agriculture and forestry can either be classified as a waste of non-waste. If the material has an existing productive use then it is considered to be a co-product or a by-product.

The Renewable Fuel Agency (RFA) classifies wastes and co-products as products with an inelastic supply in response to demand (RFA, 2010), meaning that the production of one is dependent on the production of another product (Bauen et al., 2008). Usually the production system only exists because there is one main product, or the determining product (e.g. product 'A' in **Figure 2-11**), that is in demand. In some very rare cases, there is demand for both products; so that either one can be the determining product. For example soy meal was originally a co-product from soy oil production, but increasing demands for soy meal in the agricultural sector means that this is sometimes the main driver for soy bean cultivation (Garnett, 2009).

The RFA has introduced an additional classification of a by-product (**Figure 2-11**). By-products are identified as 'production residues that are not a waste', whereas co-products are defined as 'any of two or more products coming from the same unit process or product system' (Brander et al. 2009b). A by-product is classified as so if it represents less than 10% of the farm or factory gate value, and if more it is classified as a co-product (RFA, 2010). With by-products it is assumed that its consumers have "little influence on the sustainability of the production processes for the original product" and do not need to report on the sustainability of their origin (RFA, 2010).

2.5.5.2. Accounting for Co-Products

Once a product is identified as a co-product, in LCA it is necessary to decide how co-products should be accounted for. This can be performed in two main ways:

- **Awarding substitution credits to the determining product** – where a true-co-product displaces the production of a marginal good outside of the system boundaries of the study. A credit is awarded for avoided production of the marginal good.
- **Allocating upstream impacts between the determining product and the true co-products** – involves splitting emissions between co-products based on physical or other relationships.

Ekvall & Weidema (2004) state that:

> *"The life cycle model developed in a life cycle inventory analysis should be an appropriate description of the relevant parts of the technological system. What parts are relevant depends on the aim of the study"*

Therefore the method in which co-products are dealt with should be dependent on the aim of the study, and the use of the results. Unfortunately this aspect has been confused by the ISO Standards themselves (Tillman, 2000).

The ISO Standards state that allocation should be avoided by either dividing the unit process into two or more sub-processes or expanding the product system to include the additional functions related to the co-products (CEN, 2006b). If system expansion is not possible, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the underlying physical relationships between them. If this is not possible, allocation according to other relationships, such as economic value is recommended (CEN, 2006b). Hence, the ISO standards appear to favour substitution credits, then prioritise physical allocation over price allocation.

There are key differences between ALCA and CLCA with regards to how co-products are treated in LCA calculations:

- Attributional (A)LCA – should allocate impacts between co-products by either mass, energy content or price because this considers who is directly responsible for a net change in emissions due to biofuel production (Nuffield Council on Bioethics, 2011). System expansion can only be used if there is a product with an obvious displacement (e.g. exported electricity displacing grid electricity).
- Consequential (C)LCA – should perform system expansion as it increases the scope of the analysis (Nuffield Council on Bioethics, 2011).

2.5.5.3. Allocation

Allocation is a simple yet important issue in LCA, as it can be done in different ways, and can greatly affect the results (Gnansounou et al., 2009; Kaufman et al., 2010; Mendoza et al., 2008).. Allocation has been described as the major weakness of bioenergy LCA's due to the high sensitivity of the results to allocation decisions (Aylott et al., 2012), and it can seem to be an arbitrary stage of the calculations, particularly when are co-products have different functions (CEN, 2006b; Ekvall and Finnveden, 2001; Weidema 2000).

The mass and energy content of a material is readily available, consistent over time and can be easily interpreted (Ekvall and Finnveden, 2001). Few studies use allocation by mass (Menichetti and Otto, 2008) as it is possible to reduce the GHG balance of a biofuel by creating a large quantity of a low value co-product (Reijnders, 2008a). Some studies on biofuels show that allocate by energy content provide the most favourable results (Gnansounou et al., 2009; Yan and Boies, 2013). A disadvantage of allocation by energy content is that not everything has an obvious energy-content (for example, chemicals from a bio-refinery). Allocation by price can be a good representative of what drives business decision-making, and it is usually assumed that price drives production, though in reality, policy may be the main driver (Bauen et al., 2008; Reijnders, 2008a). It can be argued that price allocation is more consistent in a regulatory context as it implies the need for industry to take ownership of GHG emissions based on economic relationships (Aylott et al., 2012). A disadvantage of allocating by price is that the economic value of products is not constant (Reijnders, 2008a) and economic values do not represent the environmental dimensions of economic activity (Pelletier and Tyedmers, 2011).

One potential complication of allocation is that this must occur at the break-off point of creation (Kindred et al. 2008a). For example, the price used to allocate between straw needs to represent be the cost of straw on the field and grain in the combine harvester (**Figure 2-12**). Data on costs is not usually provided in this fashion; therefore it is necessary to develop a shadow price of that product at the desired point (Kindred et al. 2008a).

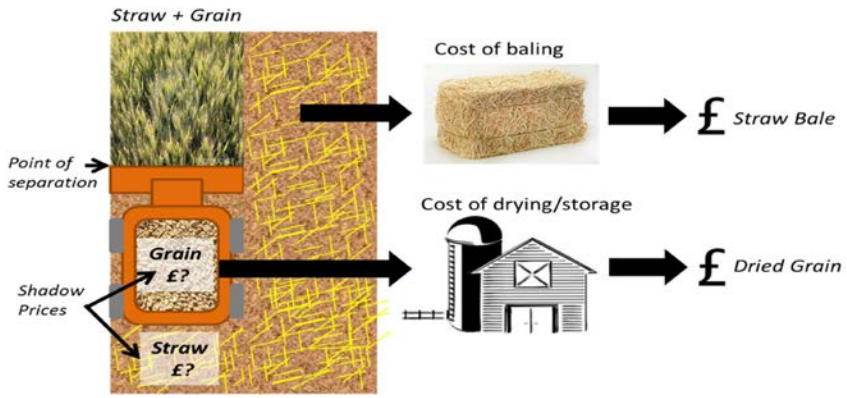

Figure 2-12. An example of calculating the shadow price with wheat grain and straw. The shadow price is the price of grain and straw at the point of separation and can be estimated by deducting the cost of grain drying and baling from the market price.

2.5.5.4. System Expansion and Substitution Credits

System expansion is considered to be applicable to most studies (Ekvall and Finnveden, 2001), and presents a more accurate approach to dealing with co-products (Bauen et al., 2008; Yan and Boies, 2013). With system expansion, instead of allocating, the final product is deducted with 'substitution credits' for each co-product. These represent the avoided emissions from substituting existing products with the co-products. Substitution credits are based on displacement of marginal goods as it is assumed that they are readily displaced when there is a substitute product on the market (Sanchez et al., 2012).

System expansion assumes that some sort of change has taken place, and confirming this has occurred is an important step in determining whether a substitution credit can be awarded. There are a few challenges of doing this: firstly, understanding the nature of supply and demand of products and alternative products will rely on some complex market, economic or even global modelling (Weidema 2000), which may not be possible, or practical. Secondly, if determinant co-products are displaced then no credits should be awarded, as they will continue to be produced regardless (Bauen et al., 2008). Thirdly, there can be many possible uses of a given product; therefore substitution credits can be rife with uncertainty (Yan and Boies, 2013).

2.5.5.5. Summary: Methodological Differences in LCA

The results of an LCA study is a result of a multitude of decisions on how the analysis should be performed (Aylott et al., 2012). The framework of a LCA study involves some key stages, all of which will differ depending on whether the study follows an ALCA or CLCA approach. In summary, there can be methodological differences between studies in the following ways:

1. ***The goal and scope*** – details the aim of the study and the scope of the assessment
2. ***The functional unit*** – specifies the final unit of measurement and the point at which the product is assessed.
3. ***Emission factors*** – different emission factors can cause differences in the results.
4. ***Co-product or waste definition*** – determines if a co-product is attributed upstream GHG emissions or whether its disposal needs to be accounted for.
5. ***Allocation procedures*** – determines the methods in which upstream GHG emissions are allocated between co-products.
6. ***System boundaries*** – specifies the sources of environmental impacts that are accounted for in the LCA study (what is and isn't included in the analysis).

It could be said that there is no right or wrong way of performing allocation in LCA studies (Kindred et al. 2008a). Different methods have their strong and weak points. A given LCA study should clearly state the allocation method used (Reijnders, 2008a). A review of LCA studies of biofuels showed that 12 out of 30 studies applied system expansion, 6 mixed their allocation methods, 4 applied consistent allocation methods, and the remaining did not allocate (Menichetti and Otto, 2008), hence there is a large array of variation in the practices being performed.

The ISO standards 14040:2006 (CEN, 2006a) and ISO 14044:2006 (CEN, 2006b), describe the main phases of performing a LCA, however they state that:

> *"There is no single method for conducting LCA. Organizations have the flexibility to implement LCA as established in this International Standard, in accordance with the intended application and the requirements of the organization"* (CEN, 2006a).

There are various options to how a LCA could be performed. Without a defined reporting methodology it is possible that two studies that examine the same functional unit could adopt completely different methodologies, and produce different results, yet both would comply with the ISO standards. The ISO standards leave a great deal of scope for interpretation and flexibility to the LCA practitioner (Aylott et al., 2011).

2.6. Current GHG Reporting Methodologies

Three GHG reporting methodologies have been adopted in policy and legislation in order to harmonise GHG assessments of products and to regulate biofuels. These include:

- The Renewable Transport Fuel Obligation (RTFO).
- The European Union (EU) Renewable Energy Directive (RED).
- The Publicly Available Specification 2008:2050 (PAS2050).

These all adopt a LCA approach in their calculation methodologies, but there are differences between them. The RED and RTFO were developed in response to concerns over biofuel sustainability, while the PAS2050 is applicable to any product or service. Although the PAS2050 is not sued for biofuel production, it is used for many cereal crops which could be used in the biofuel industry; therefore there may be overlaps or confusion in production systems. Over time we may see some of the methodologies be revised, updated and even merged, and it is possible that eventually one will "rule them all".

2.6.1. Renewable Transport Fuel Obligation (RTFO)

The RTFO was the first UK Government's policy for reducing GHG emissions from road transport. It imposes a "legal obligation on fossil fuel producers to produce or supply renewable transport fuel" and defines the basis for biofuel producers to report their GHG emissions (Black et al., 2011; RFA, 2010). It also introduces sustainability principles to consider environmental and socio-economic impacts of biofuel production. From April 2008, it was intended to deliver carbon savings of 2.6-3.0 million tonnes by 2010 through encouraging the use of renewable fuels. The saving is based on a biofuel blend of 5% by volume. The UK Biomass strategy estimated the land area required to fulfil just half of the target was in the range of 740,000 ha, based on traditional biofuel production from wheat and oilseed rape. The remaining 37.3 PJ of biofuels would be imported: split between 62% and 38% between biodiesel and bioethanol, respectively (DTI et al., 2007). Such a production volume was never achieved, and the RTFO targets were postponed due to concerns with biofuel sustainability.

The original methodology for the RTFO was written by E4Tec (Bauen et al., 2008), however it was highly modified since the publication of the Renewable Energy Directive (RED). Although the RED calculation methodology now mostly replaces that used in the RTFO this study examines the original RTFO methodology as it provides a different view on how the GHG calculations should be performed, with its own valid interpretations of the ISO Standards.

2.6.2. European Union Renewable Energy Directive (RED)

The RED (EC, 2009a) is produced by the European Parliament and the Council of the European Union as part of the Climate Change Package agreed in December 2008 (Black et al., 2011). Produced in April 2009, it promotes energy from renewable resources, and provides targets for participating Member States to commit to. The UK target is to produce 15% of all energy from renewable resources, including a minimum 10% of renewable transport fuels (EC, 2009a). The renewable energy targets do not specify the type of technology that should contribute to the energy mix, however it is envisioned that biomass will deliver about 30% of the renewable target (DECC, 2011b).

The Directive provides reporting guidelines with mandatory components which are expected to be implemented by Member States by December 2010. It includes both 'first' and 'second' generation biofuels, as well as electric vehicles. It states that biofuel production should be sustainable. The sustainability criteria are not yet fully developed, but they will ensure that biomass is not grown on bio-diverse, protected or endangered lands. Carbon released from land conversion must be included in the GHG calculations.

The RED specifies that the GHG savings from biofuels should be at least 35% before January 2017, 50% after, and 60% after January 2018 for installations that start on or after 1 January 2017. There are proposed changes to reduce the emission saving limits to 60% for plants initiating operation after 2014 (ICCT, 2012). Details are provided to how the GHG emissions and GHG savings from biofuel supply chains should be calculated. The RED requires that Member States should provide a 'guarantee of origin' for electricity and heat from biomass. These guarantees are required to prove the energy is renewable, rather than sustainable.

2.6.3. Publicly Available Specification 2050: 2008 (PAS2050)

The PAS2050 methodology (BSI, 2008b) is the first attempt to provide an applicable and consistent approach to accounting for the GHG balance from any product or service (Sinden, 2009). It was published in 2007 by the British Standards Institution (BSI) at the request of the Carbon Trust and the Department of the Environment, Food, and Rural Affairs (DEFRA) in response to a "broad community and industry desire for a consistent method for assessing the life cycle GHG emissions of goods and services". Over the last decade there has been an increase in the number of companies that have voluntarily claimed to have committed to GHG reduction strategies following PAS2050. The main driver of this is believed to prepare businesses for future carbon markets where GHG emissions are traded in the global market (Hall et al. 2010).

The main principle of PAS2050 is that the assessment uses relevant, accurate data, is complete, consistent and transparent so that the calculations are repeatable. It will allow consumers to compare similar products according to their GHG 'footprints', and facilitate the development of a 'business-to-business' database of 'foot printed' products (BSI, 2008b). Biofuel producers will not tend to apply the PAS2050 methodology to their supply chain as they are obligated to report to the RTFO, and soon to the RED, whereas PAS2050 accreditation is voluntary. The methodology is not specialised for biofuels. The PAS2050 method, however, is currently being used for food products. Wheat producers will have to measure their emissions differently depending on if they send their grain to a biofuel producer, or, say, to a bread manufacturer.

Although a number of studies have examined the effect of LCA methodology on the results, none have applied it to these specific GHG reporting methodologies that are currently in use (Gnansounou et al., 2008, 2009; Malça and Freire, 2006; Yan and Boies, 2013)**. This is identified as a gap in current knowledge and will be explored in this study.**

Even where studies mention the RED, they fail to note the specifics of the methodology, or do not adopt it correctly. For example Menichetti & Otto (2008) and Whitaker et al. (2010) mention that allocation is carried out according to energy content in the RED. Whilst this is true, there are specific rules for specific co-products (discussed in **Chapter 11**). This point is also missed by Yan & Boies (2013), who also include machinery manufacture in their system boundaries, which is explicitly excluded in the RED. Also, it is clear from the LCA carried out in Martinez-Hernandez et al. (2013) that they have not interpreted the RED correctly, as they award incorrect credits to exported electricity when it is produced from straw. Another study by Weinberg & Kaltschmitt (2013) fail to interpret the RED methodology correctly as they treat surplus electricity as a co-product and allocate by energy content. The methodologies applied in the GHG reporting methodologies are therefore more complicated and specific than simply deciding to "allocate co-products by price" (for example).

The following chapter details the research methodology in which this is assessed.

Chapter 3. Research Methodology

This chapter describes the research aims and objectives. The research aim is:

> **"To identify the relative importance of LCA methodology in biofuel GHG assessment, and compare this to the impact of variability and uncertainty."**

This chapter introduces the process by which the research aim is addressed, which is summarised in **Figure 3-1.** A more detailed account of where each stages of the analysis takes place is presented in the summary section (**Section 3.8**).

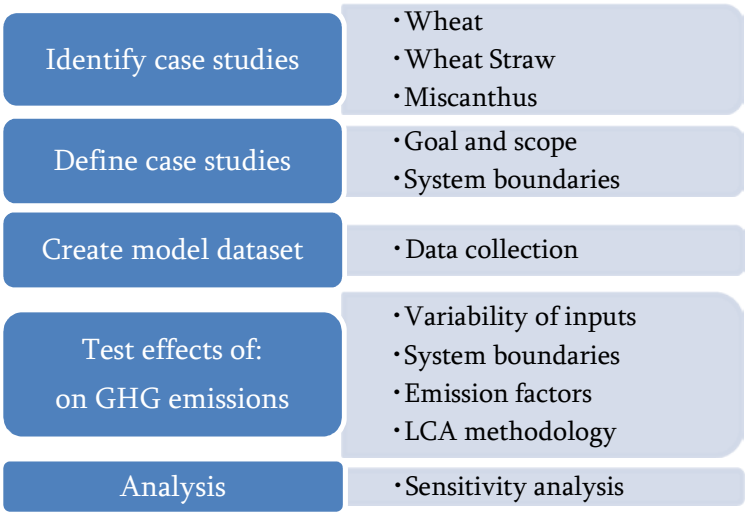

Figure 3-1. The overall research plan of this thesis.

A case study approach is used to examine the effect of LCA methodology on the GHG results. This chapter identifies these case studies and describes how the model dataset is developed. By using a case study model dataset 'real' variation due to input data is eliminated (**Figure 3-2**). The following sections provide an outline of the research plan and objectives.

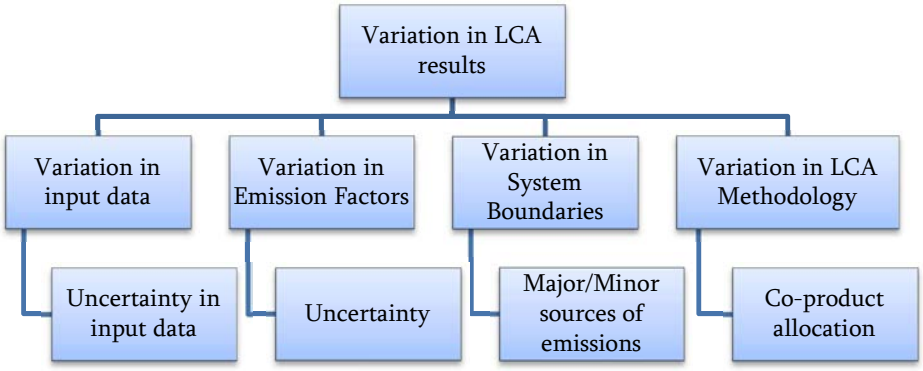

Figure 3-2. Sources of variation in LCA results studied in this thesis.

3.1. Research Plan and Objectives

The impact assessment phase of an LCA study involves converting the main processes of a production system into an environmental impact. This is a central stage of LCA and the calculations involved are crucial in determining the final result.

After the inventory data collection phase, the impact assessment involves identifying emission factors and determining the methods in which upstream GHG emissions are accounted for between co-products. **Each of these will be examined separately in this thesis, and then the final results will be combined.** First the case studies are introduced.

3.2. Case Studies

After literature review, the following three agricultural products are identified:

1. Wheat grain
2. Wheat straw
3. Miscanthus

Wheat grain was selected as the feedstock for 1st generation bioethanol because it is the most extensively grown crop in the UK (DEFRA et al., 2010), the UK is currently equipped to produce 895 million litres of bioethanol per annum (MacLeay et al., 2013) and currently the largest bioethanol plant in the UK is 'feed' wheat-based (Vivergo Fuels, 2013).

The use of lignocellulosic biomass residues is being promoted in the current European Renewable Energy Policy (EC, 2009a), which directs the current policy in the UK's Renewable Transport Fuel Obligation (DfT, 2011). Straw is a residue from arable crop production, representing a resource that can be acquired from potentially 3 million ha in the UK (DEFRA et al., 2010). The straw potential in the UK is estimated to be in the region of just under 12 million tonnes per year (Copeland and Turley, 2008).

Miscanthus, or *Miscanthus giganteus*, has been identified as the most promising dedicated energy crop in the UK (Don et al., 2012). Although historical support for establishing energy crops in the UK have helped install just over 6,000 hectares by 2013 (Natural England, 2011, n.d.), Miscanthus continues to appear as a major bioenergy candidate in current European Renewable Energy Policy. The latest projections envisage that perennial grasses will cover 25% of all land dedicated for bioenergy (EEA, 2013). It is expected that they will be used for heat and power production (EEA, 2013), although it is possible that they could be used for advanced biofuel production (Brosse et al., 2012). Wheat straw and Miscanthus have similar sugar compositions, and can theoretically provide comparable yields of bioethanol (EERE, n.d.). Technological advances are needed for 2nd generation to be taken up in UK, as although some companies have achieved commercial scale production (Woods and Bauen, 2003) there are no fully commercial lignocellulosic plants in operation currently.

A number of studies have examined wheat grain conversion to bioethanol (see Chapter 8), however very few transparent studies exist on straw-based bioethanol, and no specific studies have been performed on Miscanthus to bioethanol. **This study therefore presents a novel study of the conversion of Miscanthus to bioethanol.**

3.3. Examining the Variability of Inventory data

Inventory data provides a complete list of inputs and outputs to the product system studied. This can involve quantities of fertilisers, pesticides, farm operations or energy required to produce the selected crop. The analysis will examine how the variability and uncertainty in inventory data will affect the final result. This will be examined first without considering the effects of LCA methodology on the GHG emission results.

It important to ensure the data collected represents the current situation in the UK, because these products are being assessed to determine whether GHG emissions are being saved compared to using conventional fossil fuels. The RED has set specific emission saving targets in place, and this study will highlight where emission saving targets are met, or to identify the main limitations of the suggested approach.

Finally, a sensitivity analysis will indicate which parameters the results are most sensitive to. A sensitivity analysis provides some indication of the influence of the most important assumptions in a LCA study (Goedkoop et al., 2010). This will indicate:

- The most sensitive parameters
- Which parameters must be included in GHG assessments
- Why there is variation between studies
- How methodologies and tools can be harmonised

3.4. Examining Emission Factors and System Boundaries

The aim of this assessment is to examine how the **system boundaries** of tools may vary, and how they affect the final GHG emission results. The study also highlights how inconsistent **emission factors** can also have an impact on the GHG emission result. In this case, the effect of variation in input data is not observed as a consistent set of inventory data will be used.

There are a number of GHG calculation tools that examine the GHG emissions from producing wheat. The system boundaries will in each tool will be examined by entering the inventory data into these tools and comparing the results.

3.5. Examining the Effect of LCA Methodology on GHG Emission Results

A number of studies have examined the effect of LCA methodology on the LCA results; however they do not follow the current methodologies adopted in current UK energy policy. **This study therefore presents a novel approach to examining LCA methodology in the context of renewable energy and agricultural policy.**

A 'GHG reporting methodology' is defined as a prescribed method for performing LCA. It details key stages in the calculations, including where to draw the system boundaries and how to account for the production of multiple products in the system (Aylott et al., 2012). The LCA methodology applied should refer back to the goal and scope of the study (Nuffield Council on Bioethics, 2011), therefore in theory, the LCA method could be determined by considering the aim of the study. Here the following three steps of logic are used:

1. **Identifying the goal and scope of the GHG methodologies**
   Identify why the GHG methodologies were developed and what they seek to find out.
2. **Consider this goal and scope and identify a suitable method of applying LCA**
   Involving a review of LCA methods and use the information to identify an appropriate method in which to fulfil the identified goal and scope.
3. **Compare the two approaches**
   The 'appropriate' method will be compared to that specified in the methodologies to see if there are any inconsistencies.

The purpose of this process is to critically review the way that LCA is adopted in current GHG reporting methodologies. The process will identify the various methods that are either currently adopted in policy and GHG regulation, or potentially adopted due in future standards. It will also identify whether current GHG regulatory methodologies fulfil their original goal and scope.

The model dataset will be used to compare the results generated by each methodology. The calculated GHG emission savings and the overall GHG emissions will be compared across methodologies. This process is performed to help identify:

- The main methodological decisions that influence the results
- Why there is variation between studies
- How methodologies and tools can be harmonised

The relative impacts of both parameter sensitivity and methodological sensitivity are compared.

3.6. Uncertainty Analysis

A review by Björklund (2002) showed that Monte Carlo was the preferred type of uncertainty analysis. The methodology described in Guo & Murphy (2012) is followed. This involves estimating the probability density function (*pdf*) of each parameter. The *pdf* can follow a number of patterns, including uniform, triangular or normal (**Figure 3-3**, Goedkoop et al. 2010). The methodology in which the *pdf* is determined is listed in **Appendix 7**.

The *pdf* is deduced from data collected from growers, statistics, growers' guides and literature. For 'uncertain factors', minimum and maximum estimates are deduced and it is assumed that there is an equal chance, or uniform chance (Goedkoop et al., 2010), of the parameter falling between the two extremes. The Monte Carlo Analysis then cross-examines these ranges over multiple likelihoods to generate a probability density matrix of the final GHG result. The analysis is performed with 1000 runs. The results are presented by showing the interquartile range and the 5% and 95% percentiles.

	Range/Uniform	Triangular	Normal
Data Needed	Min and max value	Min and max value and best guess	Average and standard deviation
Graphical representation			

Figure 3-3. Example of uniform, triangular and normal distributions used in the uncertainty analysis.

3.7. Summary of Research Techniques and Objectives

The following research objectives are identified to address the main research aim:

1. **Develop a representative case study dataset in which to base LCA studies**
   The case study datasets must account for main inputs, such as the energy requirements and material use, required for production of the selected feedstocks. The dataset must also be representative in order to determine whether biofuels have reached the GHG emission saving targets.
2. **Assess the sensitivity to parameters in LCA's of biofuels and agricultural products.**
   Perform a sensitivity analysis to identify the relative sensitivity of the results to methodological changes compared to variation in parameters.
3. **Identify impacts of system boundaries and emission factors on GHG results**
   Examine the GHG results generated from a consistent dataset with different system boundaries and emission factors applied.
4. **Examine existing LCA methodologies used in GHG reporting methods.**

A desk-based review is performed to identify existing GHG reporting methodologies. The LCA methodologies are assessed to determine whether they are appropriate for their original goal and scope.

5. **Test the impacts of LCA methodology on the results**
   Use the case study model dataset to assess the effect of different LCA methodologies on the results.
6. **Combine results of the analyses of variability and uncertainty, system boundaries, emission factors and LCA methodology.**
   Identify the major cause of variations in the results, whether this is due to LCA methodology, uncertainty or variation in inputs and outputs.

In undertaking this process a number of research techniques and analytical approaches are used to perform the research objectives. These are:

- Selection of Case studies
- Selective life cycle assessment (LCA)
- Variability and Uncertainty analysis
- Sensitivity analysis

Table 3-1 summarises the research objectives, adopted research technique and the chapter in which these are addressed. The final results are discussed in Chapter 12.

Table 3-1. Summary of the research objectives, the techniques used and where these are addressed in the thesis.

Research Stage	Details	Chapters	Research Technique
Create Model Dataset	• Range data will enable analysis of variability and uncertainty	• Introduced: 2, 3 • Analysis: 6, 7 & 8	• Desk-based review • Data collection
Analysis of effects of variability and uncertainty on the GHG emission results	• To identify whether differences are mainly due to uncertainty, variability or methodology	• Introduced: 2 • Results: 9	• Selective LCA • Monte Carlo Analysis • Sensitivity analysis
Assess impacts of system boundaries and emission actors	• To identify relative importance of system boundaries • To test for sensitivity of results to GHG emission factors	• Tools identified: 2 • Analysis: 10	• Comparison of tools • Sensitivity analysis
Review LCA methodology used	• Analyse goal and scope and LCA methodology of GHG reporting methodologies	• Methods identified (5) • Analysis: 11	• Desk-based review • Selective LCA

Chapter 4. The Case Studies

This chapter defines the case studies on which the model datasets are based. First, an overview of the case studies is presented, then the goal and scope of the LCA studies is determined and the system boundaries are defined. Finally, the inventory collection phase is described. The case studies include two different crops: wheat and Miscanthus, producing two products: food and bioethanol (**Figure 4-1**).

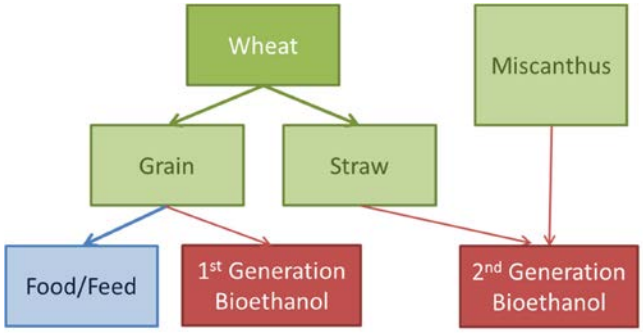

Figure 4-1. The crops and products studied.

4.1. Crop Cultivation LCA's

In **Chapter 2**, **Section 2.5** it was identified that for regulatory purposes, an attributional-based (ALCA) approach is more appropriate than a consequential-style LCA (CLCA). Therefore, for consistency, the LCA of crop cultivation will also follow an attributional (ALCA)-based approach to LCA. The goal of the crop cultivation LCA's is to therefore to examine the GHG emissions that are a direct result of producing a given unit of product to a specific point. The functional unit is "the cultivation and harvest of one hectare of wheat, wheat straw or Miscanthus with the product ready to be delivered to a consumer". The final unit of measurement is the total GHG emissions (kg CO_2 eq.) per hectare of cultivated land.

The system boundaries include both direct and indirect sources of GHG emissions. Direct GHG emissions include those that are released onsite, such as combustion of diesel in the farm machinery. Indirect GHG emissions are caused from the production of electricity and materials that are consumed onsite. They also include sources of biogenic and non-biogenic GHG emissions which can have an important role in accounting as sequestration of biogenic carbon in biomass can be released when the biomass is combusted for food or energy. Here the GHG emissions are cancelled out and are considered to be 'carbon neutral' (shown in the green-dotted boxes in **Figure 4-2**). There can also be longer-term storage of carbon in the organic component of soil (SOC) (Kochsiek and Knops, 2012; Lal, 2008a). Losses can occur due to oxidation of SOC after tillage of land (Lal, 2004a).

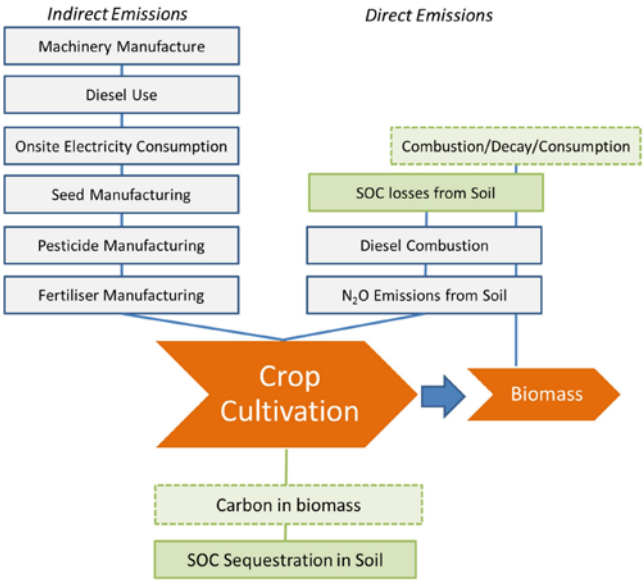

Figure 4-2. The system boundaries of the generic crop cultivation system.

There is no percentage cut-off criterion applied in the data collection. A cut-off criterion is a method of establishing whether an input or output is 'material' in a LCA (CEN, 2006a, 2006b). Instead, as much information as possible is collected, and where information is limited the compensating assumptions are stated. The following stages are excluded from the study:

- Manufacture of manure: It is assumed that manure is a waste-product from animal husbandry; therefore this material is assumed to be collected from a local farm. Where used, a delivery distance of 10 km is assumed to represent local use.
- Other farm inputs and buildings.
- Seed propagation in wheat.

The following sections provide an overview of the crop cultivation system and specific system boundaries for the individual crops.

4.2. Wheat Cultivation

Wheat is the most widely grown crop in the UK, with a total coverage of 1,939 thousand hectares in 2010, representing 32% of the total cropped area (DEFRA et al., 2010). All arable crops are grown in a similar way, with burdens arising from crop protection, fertilisation and harvesting stages (Audsley et al., 1997).

Wheat tends to be grown on heavier land therefore **ploughing** is generally necessary to loosen the soil and control weed growth (Richards 2000). This intensive method of cultivation may be performed once a rotation, or shared between rotations. It involves turning over the top nine inches of soil, burying surface debris, loosening the soil and breaking up weeds (UK Agriculture, 2012a). There are various types of plough; some disturb the soil less than others.

There are usually one or two other primary cultivations between ploughing and drilling (Richards 2000). These include **discing** and **power harrowing**, which even out clumped earth to leave a fine tilth to improve the suitability of the soil for seed planting and establishment. These are usually followed by **planting** and **rolling** (Williams et al. 2006), to increase soil contact with seeds, helping to control moisture and nutrient availability (UK Agriculture, 2012a). Applications of **fertiliser, pesticides and lime** are then made. Applications of P_2O_5 and K_2O are applied at establishment (Richards 2000) whereas N is usually applied between March and June after tillering to encourage the growth of the main leaves and grain (HGCA, 2008).

Harvesting of cereal crops usually takes place in July to early September. Traditional combine harvesters cut the straw 9 inches from the ground, thrash the wheat to separate grain from the straw, then clean the grain and deliver it to a storage hopper (UK Agriculture, 2012a). Grain drying is recommended to prepare it for storage (HGCA, 2008). Some harvesters chop the straw component so that it can be more easily incorporated, though this requires more fuel to perform (Glithero et al., 2012). Straw **baling** is carried out by a specialist machine that compresses it into blocks and secures them with string (UK Agriculture, 2012a).

4.2.1. System Boundaries of the Wheat Cultivation System

Figure 4-3 describes the specific system boundaries of the wheat cultivation system. It shows sources of both direct and indirect, fossil/non fossil (blue borders) and biogenic GHG emissions (green borders). Biogenic storage of carbon in the plant is not included because it is assumed that, within a short time-frame, the biomass will be either consumed as a food or energy resource. The system boundaries also include SOC losses from soil that may occur under wheat cultivation. SOC is discussed in great detail in **Chapter 5**, **Section 5.1.3** as it will depend on the previous use of land and how straw is managed.

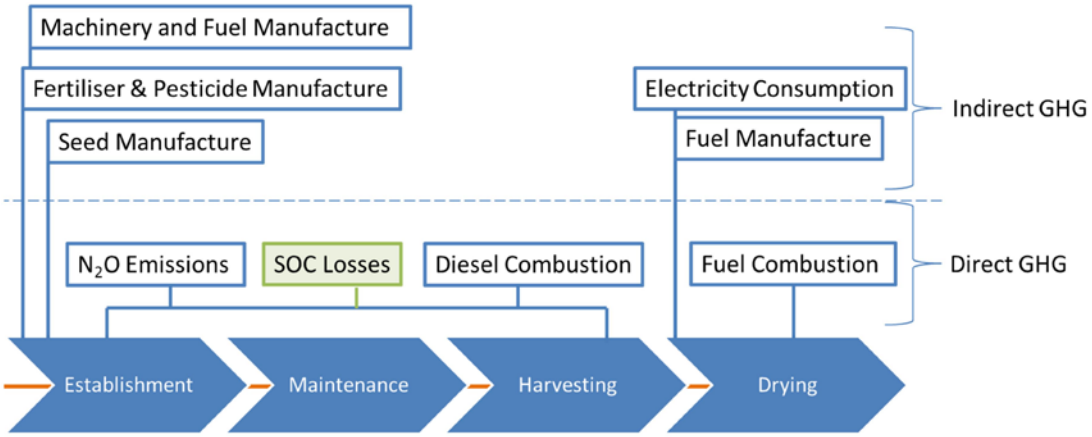

Figure 4-3. The specific system boundaries of the wheat cultivation system.

4.3. Miscanthus Cultivation

Miscanthus, or *Miscanthus giganteus*, has been identified as the most promising dedicated energy crop in the UK (Don et al., 2012). It has demonstrated high yields in trials compared to other grasses (Riche, 2005), low GHG emissions from cultivation (Hillier et al., 2009), high nutrient use efficiency (Cadoux et al., 2012) cold tolerance (Farage et al., 2006), resistance to pests and diseases (Riche, 2005), and, as a C_4 grass, it is likely to utilise water more efficiently than C_3 plants, such as reed canary grass or willow (Hall 2003; Richter et al. 2008).

The Miscanthus cultivation process involves:

- Site preparation
- Establishment of crop
- 1st Year maintenance
- Harvesting
- Termination

4.3.1. Establishment of a Miscanthus Crop

The establishment phase is the most critical phase for successful development of a perennial crop (Don et al., 2012). This can be performed in one of 3 methods:

- Seed
- *In vitro* propagation
- Vegetative reproduction by rhizomes

Establishment of **seed** is often limited due to climatic conditions, and some species are sterile (Atkinson, 2009). Seeding varieties are generally not desired as Miscanthus is not native to the UK and there are concerns that non-sterile plants would invade natural areas (Jørgensen, 2011).

Lewandowski et al. (1995) presented what may be considered to be the first transparent LCA study of Miscanthus cultivation involving ***in vitro*** propagated plantlets. The process involves placing excised auxiliary buds into a sterile medium to encourage root formation to form small plantlets (Atkinson, 2009). After 6-8 weeks establishment in a greenhouse they are planted in the field (Lewandowski et al., 1995).

Bullard & Metcalf (2001) provided the first transparent UK-specific account of Miscanthus cultivation that was established via **rhizomes**. These are the below-ground units of Miscanthus. Upon planting, the rhizomes produce both roots and shoots from growing buds (**Figure 4-4**). Year over year the rhizomes grow new terminal buds that can be broken off to form new rhizomes that can be multiplied for creation of new stock. The process can be expensive as it requires three to five years to build a sufficient rhizome population in the growing stock (Christian et al., 2005). It is still the most practical method of establishment and therefore current commercial stands of Miscanthus are established this way (Atkinson, 2009).

Figure 4-4. A rhizome showing 4 apical buds from which shoots will grow.

The process of rhizome multiplication is therefore an additional process of Miscanthus cultivation. A review of literature shows that relatively few studies include this stage. Bullard & Metcalf (2001), provide the most transparent account of rhizome multiplication. Others resources either assume an input for 'rhizomes' whose GHG emissions are calculated elsewhere (Fazio and Monti, 2011; Felten et al., 2013; Smeets et al., 2009), or do not refer to them at all (Gilbert et al. 2011; Styles et al. 2008; St Clair et al. 2008). **This research therefore provides some updated insight into the rhizome harvesting process that is relevant to a commercial stand.**

4.3.2. Rhizome Propagation

In the model case study for wheat, the process of seed production was not examined because many LCA studies have demonstrated that seed production is a negligible source of GHG emissions (Bauen et al., 2008). In contrast, literature indicates that rhizome cultivation contributes 8% of GHG emissions (AEA Technology and North Energy Associates, 2010) and 71% of the total cost of Miscanthus establishment (Calu, n.d.).

Performing an LCA of seed or rhizome production is difficult, as there is an obvious 'chicken and egg' situation that to produce rhizomes, rhizomes are required. As rhizomes can be multiplied, the origin of the parent rhizome can be uncertain. Therefore some assumptions must be made. As the most commonly planted species of Miscanthus (*Miscanthus giganteus*) is sterile hybrid of *Miscanthus sinensis* and *Miscanthus sacchariflorus* it is probable that the original rhizome stock would be produced in a controlled environment through cross breeding. This study assumes that rhizomes can be collected from natural stock and a sensitivity analysis is performed to test the importance of this assumption.

4.3.3. Miscanthus Cultivation

When cultivating a Miscanthus crop, rhizome planting occurs in the spring, and by early summer the plant would have reached a height of 1-2 metres (DEFRA 2001). Herbicides are applied during this phase, and then again after some harvest events. It is conventional practice to leave the plant to grow for 2 full years until the first harvest is made, though some studies suggest mowing, or 'topping', the 1st year's growth (Gilbert et al. 2011).

Harvesting is carried out using a forage harvester and the cut biomass is baled. The forage harvester can be modified to chip the biomass for equestrian use (M. Carver pers. com. 2012). During the 1980's, harvesting was carried out according to traditional hay making, which is in July and August. It was found, however, that the process of drying and storage the biomass meant that it was impossible to guarantee a fuel that was of sufficient quality at a suitable cost (Hadders and Olsson, 1997). Harvesting after the winter period was identified as more suitable option for biomass combustion as the crop has a lower moisture and nutrient content and is of better quality (Hadders & Olsson 1997; Lewandowski & Kicherer 1997).

Harvesting of the regrowth is repeated annually until the final termination event, during which the crop is finally harvested and destroyed. Termination involves soil tillage and application of herbicides (Bullard and Metcalf, 2001).

4.3.4. System Boundaries of the Miscanthus Cultivation System

Figure 4-5 and **Figure 4-6** describe the specific system boundaries of the rhizome and Miscanthus cultivation systems. They show sources of both direct and indirect, fossil/non fossil (blue borders) and biogenic GHG emissions (green borders). Biogenic storage of carbon in the plant is not included because it is assumed that, within a short time-frame, the biomass will be consumed for energy production.

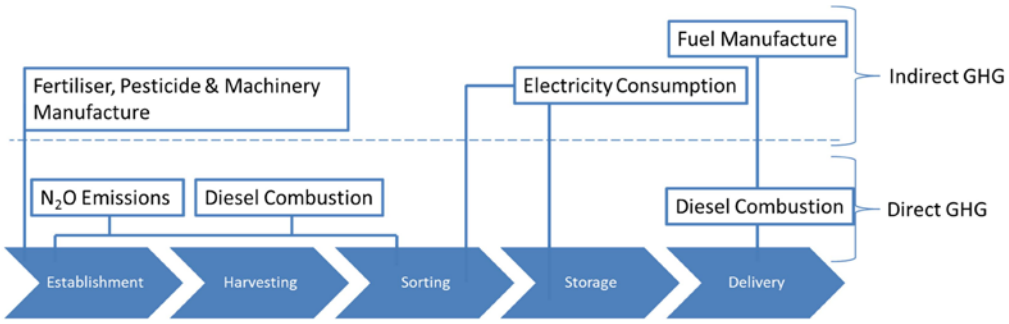

Figure 4-5. Specific system boundaries of the Miscanthus rhizome cultivation system.

The rhizome study feeds into the main crop Miscanthus LCA. The rhizome propagation phase does not include changes in SOC because it is only established for between three to five years. The main crop Miscanthus system boundaries include SOC sequestration in the soil under the crop (**Figure 4-6**). SOC is discussed in great detail in **Chapter 7**, **Section 7.4**, as it will depend on the previous use of land and the harvest time.

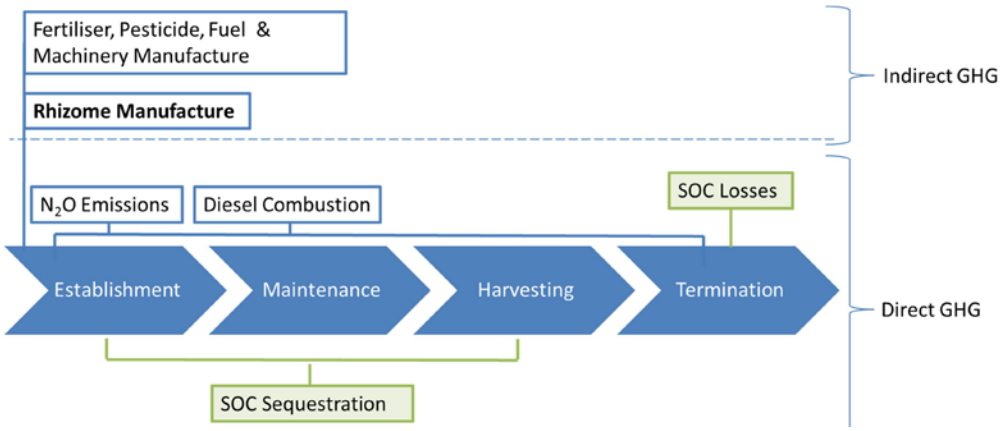

Figure 4-6. Specific system boundaries of the Miscanthus cultivation system.

4.4. Bioethanol Production

There are two main routes to bioethanol production: via gasification or fermentation. **Gasification** is a highly feedstock-flexible and advanced technique of traditional charcoal manufacture, where the biomass material is heated to high temperatures within an air-controlled environment. The result is a gaseous mixture of hydrogen, carbon monoxide, methane, ethane, propane, butane and char (Agrawal, 1987). The gas can then be converted to bio-methane via catalytic methanation, or to bioethanol via microbial fermentation with micro-organisms (Keshwani, 2009).

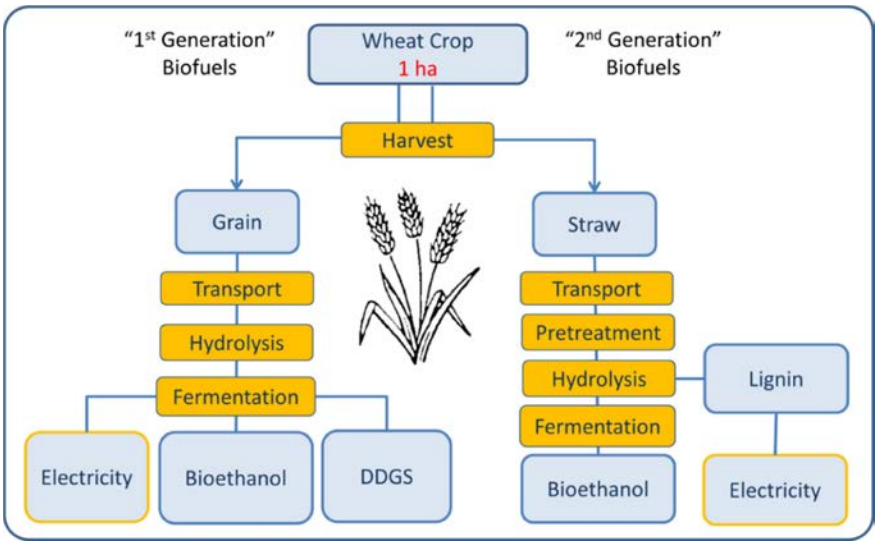

Figure 4-7. The fermentation routes of 1st and 2nd generation bioethanol production from a wheat crop (DDGS = dry distillers grains and solubles).

The **fermentation** route is explored in this study, as it is adopted in current bioethanol plants. Bioethanol can be produced from wheat grain (1st generation), crop residues or other lignocellulosic material (2nd generation, (**Figure 4-7**)). The goal of the full bioethanol ALCA studies is to examine the GHG emissions that are a direct result of production of "1 MJ bioethanol from wheat grain, wheat straw or Miscanthus that is produced to the factory gate". The final unit of measurement is the total GHG emissions (kg CO_2 eq.) per MJ.

A fossil fuel comparator is required in order to assess the relative GHG emission savings of the full bioethanol assessment. The comparator to bioethanol is gasoline as this directly replaces its use. The GHG emissions from bioethanol are compared with the idealised GHG emissions from producing and combusting one MJ gasoline (estimated at 83.8 g CO_2 eq. MJ^{-1} (Biograce.net, 2012)). The CO_2 emissions from combustion of bioethanol are considered to be biogenic however the non CO_2 GHG emissions (methane (CH_4) and N_2O) are included. The emission rate is 0.04 kg CH_4 and 0.007 kg N_2O MJ^{-1} (AEA Technology and North Energy Associates, 2010).

The system boundaries of the bioethanol LCA's will include inputs of biomass, energy, chemicals, and outputs of bioethanol, electricity and other co-products. These are shown in **Figure 4-8**.

As with the crop LCA studies, there is no percentage contribution cut-off criterion applied and as much information as possible is collected. Where data are limited the compensating assumptions are stated. The following stages are excluded from the study:

- Bioethanol plant manufacture: Plant manufacture is excluded in the LCA due to lack of data and evidence that this is relatively small over the lifetime of the plant (AEA Technology and North Energy Associates, 2010).
- Carbon dioxide emissions from bioethanol combustion: are assumed to be zero, but CH_4 and N_2O are accounted for according to literature (Elsayed et al., 2003).
- GHG emissions from minor chemicals used in processing, due to lack of data and small quantities being used.

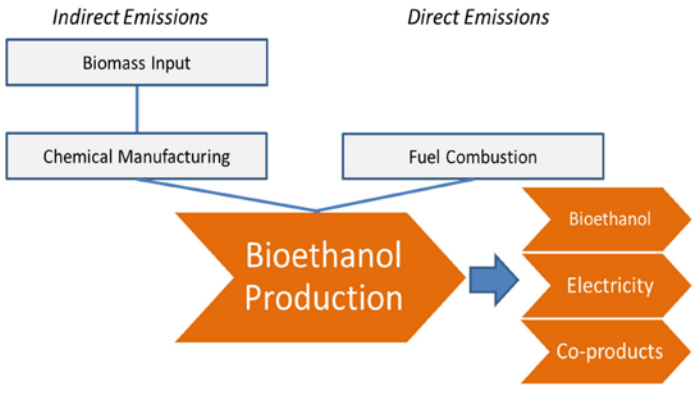

Figure 4-8. System boundaries of bioethanol production system.

4.5. The 1st Generation Bioethanol Production Process

The 1st generation production system is described by Clarke et al., 2008; Malça and Freire, 2006, as well as in some online tools that calculate GHG emissions from biofuels. These include BEAT (AEA Technology and North Energy Associates, 2010), the HGCA Bioethanol Calculator (HGCA, 2011), Biograce (Biograce.net, 2012), the Roundtable of Sustainable Biofuels Tool (RSB, 2012) and the Renewable Fuel Association's Solid and Gaseous Biomass Calculator (Westphal et al., 2011).

The fermentation process utilises sugars and starch as a feedstock. Sucrose is the easiest feedstock to utilise as the 6 carbon sugars are readily available for conventional fermentation technology to produce bioethanol (Keshwani, 2009). Starch is a slightly more complex molecule that must be saccharified to simple sugars through acid or enzymatic hydrolysis. Wheat is an example of an ethanol production pathway in the UK (Yan and Boies, 2013).

The conversion process includes transportation of grain, milling, hydrolysis and fermentation, distillation to bioethanol, and final distribution to the pump for consumption. Milling is required to increase the surface area of the wheat grain.

The hydrolysis or saccharification phase involves creating a mash with the grind by adding water and introducing enzymes to facilitate the breakdown of starch. Fermentation leads to the production of ethanol at a 10-15% concentration, and after is distilled to 95% for fuel bioethanol (Malça and Freire, 2006). The resulting bioethanol is typically dehydrated further and then blended with conventional gasoline in either low (5%) or high (85%) concentrations (Hammond et al. 2008).

The following inputs are identified in the 1st generation bioethanol production system.

- **Wheat grain** - provides the starchy substrate
- **Wheat straw** - (if used) required for heat and power generation
- **Natural gas** - (if used) required for heat and power generation
- **Enzymes** - α amylase and amylo-glucosidase are required for conversion of starch to glucose (Kelsall & Lyons 1999)
- **Yeast** – *S. cerevisiae* is the most commonly used yeast for fuel ethanol production (Tomás-Pejó et al., 2012). Required for fermentation.
- **Caustic soda** -detergent for cleaning equipment (Kelsall & Lyons 1999)
- **Diammonium phosphate** – a nutrient source for yeast (Tomás-Pejó et al., 2012)
- **Sulphuric acid/Phosphoric acid/Calcium chloride** – required for regulation of pH (AEA Technology and North Energy Associates, 2008; Bernesson et al., 2006)

Wheat grain contains 60-70% starch, which forms the feedstock for the fermentation process. The remaining protein and fibre components cannot be converted to bioethanol (Weinberg and Kaltschmitt, 2013). These residues form a co-product known as dry distillers grain and solubles (DDGS). As of January 2008, DDGS can be dried and sold as animal feed or burnt for

energy (Environment Agency, 2008). It is possible to obtain more valuable products such as bran and gluten by processing prior to the hydrolysis phase (Weinberg and Kaltschmitt, 2013), however this is not explored here as no bioethanol plants are known to practice it (Clarke et al., 2008).

Heat and power is required for processing, and is usually provided using a combined heat and power (CHP) plant. The plant can be fuelled by natural gas or by biomass. In some cases in the literature (AEA Technology and North Energy Associates, 2008; Mortimer et al., 2004; Punter et al., 2004; Yan and Boies, 2013), there is excess electricity to that required by the process, and this can be exported to the national grid. The ratio of heat and power produced will depend on the technology used, which may be limited by economic factors (Punter et al., 2004).

Figure 4-9 shows the specific system boundaries of 1st generation bioethanol production. They include inputs of biomass, energy, chemicals, and outputs of bioethanol, electricity and other co-products. As with the crop LCA studies, the system boundaries include direct and indirect GHG emissions. All of these GHG emissions are non-biogenic.

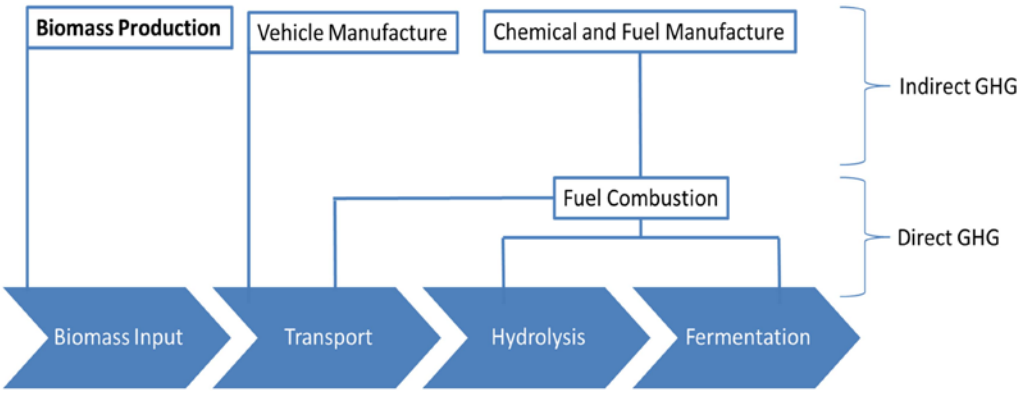

Figure 4-9. System boundaries of the 1st generation bioethanol production system.

4.6. The 2nd Generation Bioethanol Production Process

Bioethanol can be produced from lignocellulosic material via either a microbial or biochemical conversion process (Singh et al. 2010). Literature is currently limited in terms of the range of LCA inventory data available. A review of literature shows that most LCA studies derive inventory data from two technical reports produced by the National Renewable Energy Laboratory in the US (Aden et al., 2002; Wooley et al., 1999). The LCA is usually performed for corn stover and wood, however more recent studies have examined the use of wheat straw (Borrion 2012; Wang et al. 2012). Only one study has been performed on the production of bioethanol from 'energy grasses' (Wang et al. 2012), **therefore this study offers a novel assessment of a Miscanthus-specific process.**

Second generation biofuel production requires a far more complex process than for 1st generation. Lignocellulosic material is composed of cellulose (35-50%), hemicellulose (20-35%), and lignin (10-25%) which are extensively connected to create a complex, rigid structure that has adapted in order to protect the carbohydrate fraction from decay (Saha, 2004). When producing ethanol from lignocellulose materials, the main target for bioconversion is the carbohydrate cellulose: a molecule composed of at least 500 alternatively inverted D-glucose monomers that are linked by β 1-4 glycosidic bonds to form a highly crystalline, long, linear strand. Hydrogen bonds within cellulose strands stabilise into dense microfibrils that have a tensile strength comparable to steel (Alberts et al., 2002). The material therefore requires a greater deal of processing than sucrose or starch-based materials, but its abundance in low-cost sources makes it an attractive substrate for bioethanol production (Akin, 2007).

Pre-treatment is required in order to access the fermentable components of the biomass (**Figure 4-10**). Pre-treatment processes tend to include a comminution process to increase the surface area of the biomass, then treatment with either acid, steam explosion, ammonia fibre explosion, hot water (Balat, 2011), wood decay fungi (Wan & Li 2012) or microwaves (Keshwani, 2009).

Lignin is the largest non-polysaccharide-based component of lignocellulose (Kristensen 2008). It is a highly complex polymer composed of three types of phenolic acids, or monolignols: *p*-coumaryl alcohol, coniferyl alcohol and synapyl alcohol, the proportions of each differing between plant components and species (Keshwani, 2009). It is this compound that a number of valuable chemicals can be derived from, such as vanillin, phenols, syringols, benzene and toluene (etc., more in Varanasi et al. (2013)). Therefore the bioethanol plant concept could develop into that of a biorefinery (Cherubini, 2010b). Lignin is also a source of some inhibitory compounds to fermentation, such as furfural (Horvath et al. 2003).

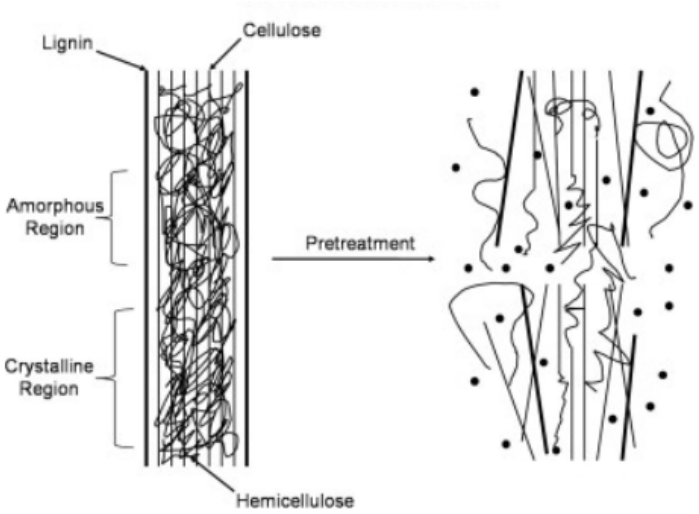

Figure 4-10. Illustrating the pre-treatment of lignocellulosic material (Mosier et al. 2005).

The following inputs are identified for 2nd generation bioethanol production:

- **Lignocellulosic material** - required as the substrate for bioethanol production
- **Acids** – used for pre-treatment of lignocellulosic material
- **Enzymes** – cellulase used to reduce cellulose to β-glucose.
- **Yeast and other enzymes** – to ferment glucose and other 5 carbon sugars to ethanol.
- **Diammonium phosphate** – a nutrient source for yeast (Tomás-Pejó et al., 2012)

The fermentation process yields a liquor of lignin and sugar residues that can be separated via centrifuge (Rabelo et al., 2011) or filter press (Aden et al., 2002). It is assumed that the lignin residue is dried and combusted for heat and power production (Aden et al., 2002; Wooley et al., 1999), though in practice the combustion properties of lignin show light to moderate fouling tendencies and the carry-over of chemicals from pre-treatment phases may be detrimental to the combustion system (Blunk and Jenkins, 2000). Theoretically however, it is expected that the lignin produced can supply the heat and power demands for the conversion process (Aden et al., 2002; Slade et al., 2009). Like the 1st generation bioethanol system, this can also lead to the production of excess electricity.

The renewable chemicals that can be derived from lignin are not included in this assessment due to a lack of data on their relative yields. Small quantities of waste are also produced as sludge and gypsum (Borrion et al. 2012). It is assumed that these are sent to landfill.

Figure 4-11 shows the specific system boundaries of 2nd generation bioethanol production. They include inputs of biomass, energy, chemicals, and outputs of bioethanol, electricity and other co-products. As with the crop LCA studies, the system boundaries include direct and indirect GHG emissions. All of these GHG emissions are non-biogenic.

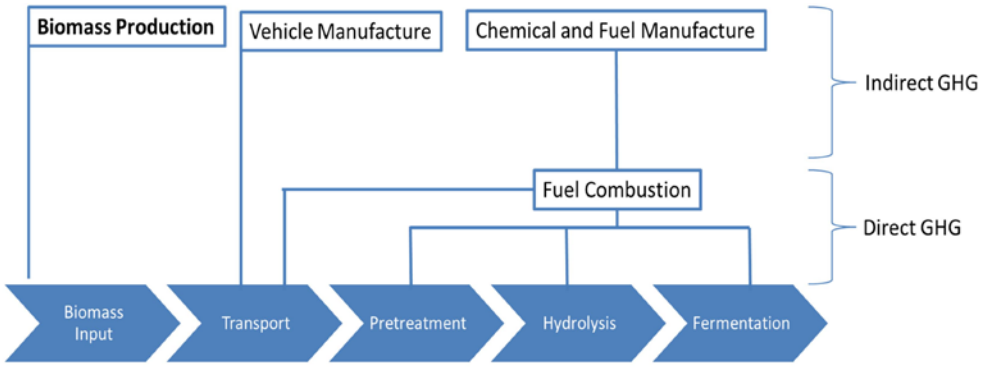

Figure 4-11. System boundaries of 2nd generation bioethanol production.

4.7. Summary of Case Studies

This chapter provides an overview of the four case studies examined. The two crops examined are wheat and Miscanthus. These are selected as one leads to the production of two or more products, and one is a single-output crop. Also one crop is annual, whereas the other has a lifetime of up to 20 years.

A number of inputs are required for crop cultivation and bioethanol (**Table 4-1**). Crop cultivation usually requires establishment, the consumption of fertilisers and pesticides and diesel to fuel machinery for processes such as ploughing and harvesting. Bioethanol production requires the consumption of fuel and chemicals and enzymes for processing.

Table 4-1. Summary of inputs and outputs to the case studies.

Case Study	Material Inputs	Material Outputs
Wheat Cultivation	• Fertiliser • Pesticides • Seed • Fuel • Machinery	• Grain • Straw
Miscanthus	• Rhizomes • Fertiliser • Pesticides • Fuel • Machinery	• Rhizomes • Miscanthus
1st Generation Bioethanol	• Biomass • Chemical reagents • Biological reagents • Fuel	• Bioethanol • DDGS • Electricity
2nd Generation Bioethanol	• Biomass • Chemical reagents • Biological reagents • Fuel	• Bioethanol • (Lignin) • Electricity

Chapter 5. Data Collection

In order to perform a LCA study, a great deal of data must be collected to quantify inputs and outputs to the system, calculate the GHG emissions associated with their production, and to determine how they are split between co-products. This section describes what data is needed and where data is collected from.

Four types of data are collected in this study, and these are shown in **Figure 5-1**, where the structure of the chapter is detailed.

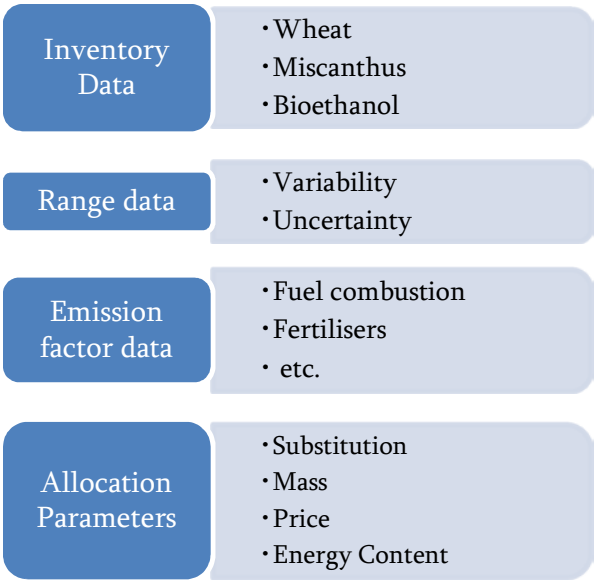

Figure 5-1. Structure of Chapter 5 showing types of data required in this study.

5.1. Inventory Data

The inputs and outputs to the model datasets must be identified and quantified. The accounting process is referred to as the **inventory analysis phase** of LCA (CEN, 2006a). To generate the inventory the data must be collected from either new or existing knowledge.

5.1.1. Wheat Cultivation Inventory Data Collection

In this study it was not feasible to directly record inventory data for wheat cultivation; however it was possible to collect some secondary data from farmers' records and completed questionnaires. An arrangement was made to use the results from a farm survey that took place in 2010/11 that recorded site inputs for 60 milling wheat crops. The survey was originally performed to examine the GHG emissions from the production of bread, which requires milling wheat, although specific details of the questionnaire are confidential. Some adjustment of the data is required to represent feed wheat, which is the preferred feedstock for bioethanol production (Smith et al. 2006).

In some instances, the quality of the data collected from farmers is poor, so literature is consulted in order to cross-reference and validate the results.

5.1.2. Miscanthus Cultivation Inventory Data Collection

There are currently few contractors that perform commercial-scale Miscanthus establishment in the UK (M. Carver pers. com. 2011). Many of the existing stands are under 10 years old. For inventory data collection, two contractors (T. Barton and I. Webber) were interviewed in the summer of 2011. Also the rhizome harvesting process was observed in the spring of 2012. From these sources it was possible to identify all operations required during both rhizome propagation and Miscanthus cultivation.

The range of diesel fuel consumption rates for each operation was collected from contractor records. For rhizome cultivation, only one commercial contractor is currently in operation. Likewise, typical pesticide application rates were collected from one contractor. The data is cross-referenced with data from literature.

5.1.3. Land-Based GHG Emissions

Three main sources of land-based GHG emissions are examined here:

- **N_2O Emissions from soil**
  - From fertiliser application (direct and indirect)
  - From crop residue incorporation
- **Land use change**
  - **Direct**

They are discussed in the following sub-sections.

5.1.3.1. Fertiliser Application

In this study, the IPCC Tier 1 approach is used to estimate N_2O emissions from soils (De Klein et al., 2006). It provides default values for 'average application' of nitrogen to soil (Bouwman, 1996; Stehfest and Bouwman, 2006). The default values are presented in **Table 5-1**. Indirect emissions of N_2O due to volatisation and leakage and specific emission factors for manure application to soil, are also included.

Table 5-1. Direct Fertiliser Induced Emission (FIE) in the IPCC Tier 1 default (De Klein et al. 2006), and the main contributing literature resources.

Resource	Average (%)	Minimum (%)	Maximum (%)
Bouwman 1996	1.25	0.25	2.25
Stehfest & Bouwman 2006	0.91	0.45	1.9
IPCC Tier 1 Default	1	0.3	3

5.1.3.2. Crop Residue Incorporation

Nitrous oxide emissions from soil also occur when nitrogen is applied in the form of crop residues. These emissions are calculated based on the IPCC Tier 1 calculation methodology. Details of the parameters used in the calculations are provided in **Appendix 4**. A sensitivity analysis indicates that the results are most sensitive to the straw yield and the emission factor per kg N (**Figure 5-2**).

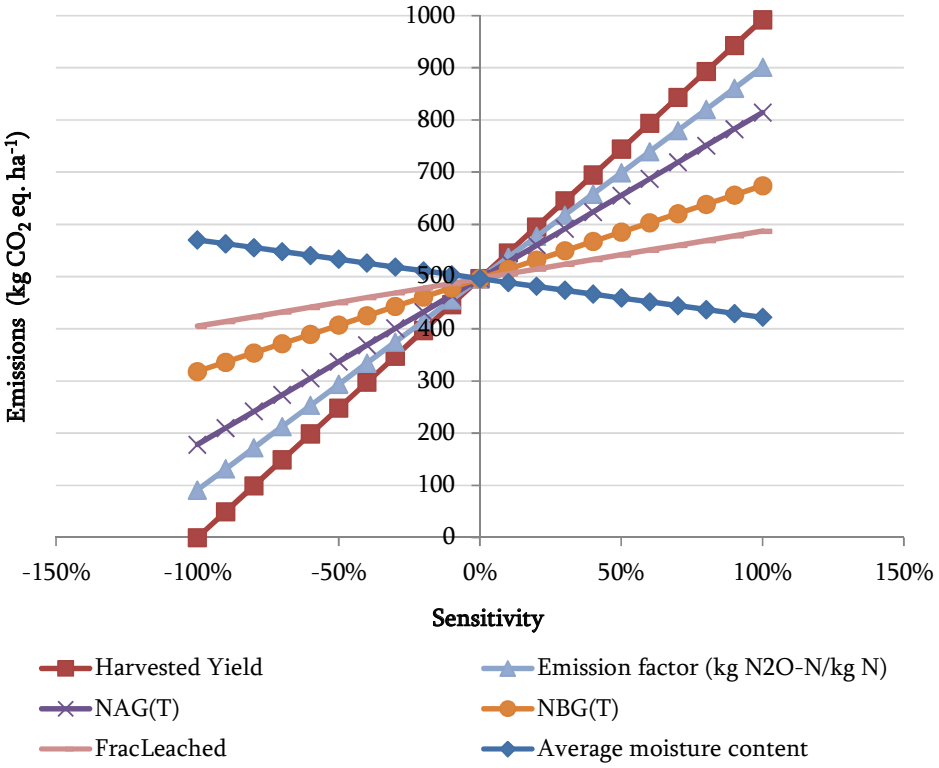

Figure 5-2 . The sensitivity of N_2O emissions from crop residue incorporation.

5.1.3.3. Land Use Change

Land use change (LUC) occurs when a given use or management of land is changed (Bickel et al., 2006). This can occur either directly (DLUC), or indirectly (ILUC) as a consequence of changes in production rates and prices of other commodities (Sanchez et al. 2012,).

The RED requires that any carbon stock change is accounted for in GHG reporting of biofuels (EC, 2009a). It provides a simplified methods for calculating DLUC in a supporting document (EC, 2010a). In the RED, GHG emissions from LUC are calculated by comparing the carbon stock per unit area before and after the LUC event. Changes in residue management also represents an example of a LUC event (Powlson et al. 2011), however there is on-going debate around the impacts of straw removal from soil (Cherubini and Ulgiati, 2010), and few studies have placed these impacts in the context of a LCA study.

5.1.4. Bioethanol Production Inventory Data Collection

Inventory data must be collected for the two conversion processes identified in the case studies. It was not possible to collect conversion data directly from bioethanol plants so it was collected from literature sources.

5.1.5. Summary: Inventory Data Collection

A combination of data sources is used during inventory data collection. These are summarised in **Table 5-2:**

1. **Grower data** – Questionnaires, interviews etc.
2. **Growers guides** – costing books and guidelines for farmers
3. **Statistical sources** – UK statistics, industry statistics
4. **Literature** – Academic literature

Table 5-2. Sources of data for various aspects of the LCA's.

Crop	Parameter	Grower data	Statistical sources	Growers' guides	Literature	Range Data
Wheat	Seeding rates	✔		✔	✔	
	Fertiliser application	✔	✔	✔	✔	✔
	N_2O Emissions				✔	✔
	Pesticide application		✔	✔	✔	
	Operations	✔		✔	✔	✔
	Diesel consumption			✔	✔	✔
	Grain drying	✔			✔	✔
	Grain yields	✔	✔	✔	✔	✔
	Straw yields	✔		✔	✔	✔
	SOC changes				✔	✔
	Nutrient off-take			✔	✔	✔
Miscanthus	Rhizomes	✔			✔	
	Fertiliser application	✔			✔	✔
	N_2O Emissions				✔	✔
	Pesticide application	✔		✔	✔	
	Operations	✔			✔	✔
	Diesel consumption	✔			✔	✔
	Miscanthus yield	✔			✔	✔
	SOC sequestration				✔	✔
	Nutrient off-take				✔	✔
Both	Transport				✔	
	Conversion				✔	✔

Grower data is used where possible as this is directly taken from farm records. Other resources are used to both validate data provided from growers and compensate for missing data.

Statistical information is available on fertiliser (Thomas 2011) and pesticide use (FERA, 2010), yields of wheat (DEFRA et al., 2010) and transport emissions (DEFRA and DECC, 2012).

Growers' guides include the Agricultural Budgeting and Costing Book (Agro Business Consultants Ltd, 2011) and the Farm Management Pocketbook (Nix, 2011). For Miscanthus specifically, the DEFRA Guideline (DEFRA, 2007) describes typical cultivation guidelines. Literature is also used to cross-reference data provided from the above resources. The IPCC calculation methodology is used for N_2O emissions from soil and direct land use change.

5.2. Variability and Uncertainty Data

An analysis of variability and uncertainty is performed in order to examine causes of variation in LCA emission results due to 'real' differences in input data, and due to 'uncertainty' in the emissions that occur due to inputs or processes (Whitaker et al., 2010).

Range data will be collected for all of the 'significant' inputs and outputs. It will include how process inputs and outputs may range spatially and temporally, and the impact this may have on GHG emission savings (Yan and Boies, 2013). **Table 5-3** indicates the sources of uncertainty and variability examined, and where the data are derived from.

Table 5-3. Details of uncertainty and variability included in this study.

Data	Aspect of LCA	Uncertainty	Variability
Cultivation	Yields and site inputs	Literature review (for crop requirements and temporal aspects)	Review of literature and statistics
Conversion	Yields and inputs	Combined in the literature review	
Emission Factors	Fuel consumption	Literature review (for operations)	Variability from growers (for operations performed)
	Manufacture of Inputs	Ahlgren et al. 2012 (for manufacture)	Variability from growers (for application rates)
	N_2O emissions from soil	De Klein et al. 2006 (for emissions)	Variability from growers (for application rates)
	DLUC	Combined in the literature review	
Allocation	Energy Content	Moisture content	N/A
	Prices	Type of crop	Temporal variability
	Substitution	Type of product displaced	N/A

5.3. Emission Factors

Emission factors are a crucial element of the **impact assessment** phase as they are used to convert the inventory data into a total emission to the environment. There are also GHG emissions from biological processes that must be accounted for. The following sub-sections describe where the GHG emission factors for the main inputs to the above processes are derived. All the GHG emission factors for these inputs can be found in **Appendix 3**.

5.3.1. Fuel Consumption

Fossil fuels which are combusted for heat or power and as a result, various emissions of environmental consequence are released, including GHG's. Emission rates for fossil fuel combustion are derived from the Biomass Environmental Assessment Tool (AEA Technology and North Energy Associates, 2010). The GHG emissions from upstream processes to provide fuel are provided from the latest DEFRA guidelines for GHG reporting (DEFRA and DECC, 2012). For consumed electricity, only the upstream GHG emissions are accounted for, as this is produced from combustion of fuels elsewhere (DEFRA and DECC, 2012).

5.3.2. Fertiliser Manufacture

The production of fertilisers demands high energy input and is responsible for a considerable proportion of GHG emissions from cultivation (Mortimer et al., 2004). This study uses emission factors for Best Available Technology (BAT) and average fertiliser production in Europe, representing minimum and maximum GHG emissions for fertiliser application, respectively.

Althaus et al. (2007) provides a review of ammonia and nitric acid production, which is used in the Ecoinvent Database. In Althaus et al. (2007), the data provided on BAT production is incomplete, as it does not include N_2O abatement. Also it does not describe the level at which N_2O emissions can be abated. Data was kindly personally provided by Frank Brentrup, from the Yara Research Centre in Hanninghof, Germany. Emissions for P_2O_5 and K_2O-based fertilisers are derived from the Ecoinvent Database. It is assumed that there is a +/- 20% certainty range on fertiliser manufacture GHG emissions (Ahlgren et al., 2012).

Organic fertilisers are a combination of composts, wastes and manures. Compost manufacture is derived from the Ecoinvent Database (EcoInvent, 2007).

5.3.3. Pesticide Manufacture

Ideally, LCA studies would perform an individual assessment of each pesticide applied within the system boundaries of their study. This is, however, rarely done as inventory data is difficult to access (it is usually confidential), there is huge range of active ingredients available in the market, and the number of licensed products changes rapidly (Sutter, 2010). When they are included in a LCA study, they are generally classified together as 'generic pesticides' or 'agrochemicals' (Elsayed et al., 2003). The Ecoinvent Database provides a list of both specific and generic pesticides.

5.3.4. Machinery Manufacture

In a LCA study, farm machinery construction is allocated to given amount of time (e.g. hours), used to carry out a given task, as a proportion of the total lifespan of the machine (assumed to be 7000 hours (EcoInvent, 2007)). This allows for the assumption that the machines are used for other activities during their lifetime.

The environmental impacts of machinery manufacture are estimated in two main ways. It can be based on the material composition of the machine, for example in Audsley et al. (1997), who assume a simple breakdown of 95% steel and 5% rubber, whereas the Ecoinvent Database (2007) and Heller et al. (2003) adopt a more detailed approach. The detailed method is time-consuming when examining a large range of equipment.

Another method is to utilise cost multipliers. These are developed using Input-Output Analysis (Hetherington, 1996) of industries and use energy consumption, emissions and overall profit to assign each an emission factor for each unit of currency old from the manufacturer to the consumer (N. Mortimer pers. com. 2010). The cost data method also includes energy inputs and emissions from constructing the machines and providing and sourcing the various parts. An example of this is provided by North Energy Associates (North Energy Associates et al., 2007).

These estimates were compared for a small 100 HP tractor, with a total weight of 4 tonnes, and typical cost of £32,000 (Nix, 2011). **Figure 5-3** shows that the emissions are over-estimated using the cost-based method. Simplifying the composition of a tractor to just rubber and steel and excluding production energy requirements means that a simple method gives the lowest GHG emission. Ecoinvent Data is used in this study, which appears to provide an estimate that falls between the cost and simple composition-based methods.

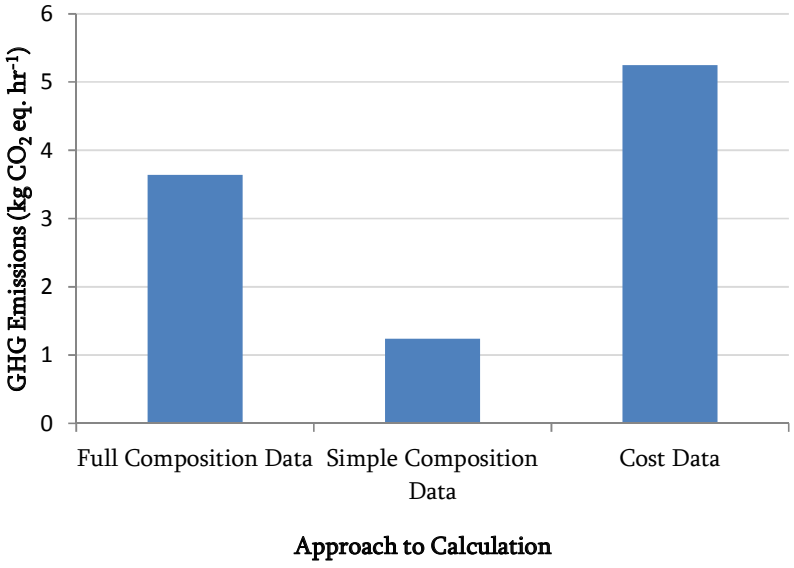

Figure 5-3. Different GHG emission estimates for 1 hour's work of a 100 hp tractor.

5.3.5. GHG Emissions from Transport

Transport distances can theoretically be determined by the scale of operation, and the feedstock density within the available harvest area (Overend, 1982), but this is not the focus of here. This process seeks to optimise the transport radius required to supply a given rate of feedstock to a given scale of plant. Transport distances between the field and the bioethanol plant are included in all LCA studies selected for study.

Transport fuel requirements are based on data provided in the Ecoinvent database. It is assumed that all transportation takes place in a 44 gross vehicle weight (GVW) truck. It is also assumed that wheat straw and Miscanthus is baled to the highest density available (MF190 at 602 kg $bale^{-1}$ (BigBale Co., n.d.)), as denser forms of biomass would be more efficient during handling and transportation.

5.4. Allocation Parameters

In order to perform either system expansion, or allocation between co-products, data is needed on:

- The mass of the co-products
- System expansion: identify suitable substitute products and the GHG emissions from their production
- Economic value: the value of the product at the point of creation
- Energy content: the energy content of the product at the point of creation

It is important to note here that the point of allocation is when the products are created (**Figure 5-4**).

Allocation should not include any processes that occur after co-product creation, including those that turn less valuable co-products to valuable ones (Aylott et al., 2012). For example, the drying of DDGS would be excluded from the system boundaries if allocation was taking place. This is particularly important as when DDGS leaves the fermenter it has a very low value as it contains a lot of water. After drying, which requires significant energy inputs; it has a much higher value. Therefore it is important to specify this, or else GHG emissions are attributed incorrectly. When system expansion is carried out the drying process would be included in the system boundaries and a credit is awarded for the finished product.

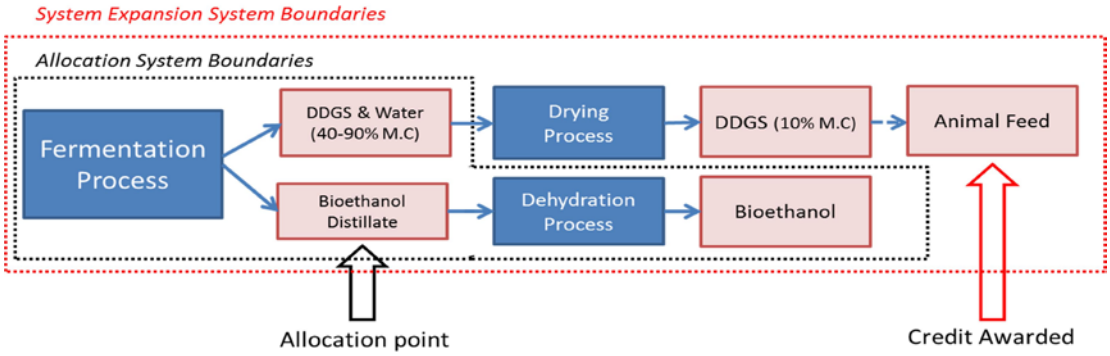

Figure 5-4. System boundaries when allocation takes place or when system expansion is carried out. Shows the point of allocation and the GHG emission credit awarded to dry DDGS.

With the exception of bioethanol and electricity, the energy content is based on the lower heating value of products at a specific moisture content (LHV_{ar}) is calculated by using the following Milne equation (**Equation set 5-1**).

Equation 5-1. Equation set required to calculate lower heating value (as arrived).

$$HHV_{ar} = HHV_{dry} \cdot (1 - w/100)$$
$$HHV_{dry} = HHV_{daf} \cdot (1 - ash/100)$$
$$LHV_{dry} = HHV_{dry} - 2.442 \cdot 8.936\ H/100$$
$$LHV_{ar} = LHV_{dry} \cdot (1 - w/100) - 2.442 \cdot w/100$$
$$LHV_{ar} = HHV_{ar} - 2.442 \cdot \{8.936\ H/100\ (1 - w/100) + w/100\}$$

Where:

$HHV_{ar/dry/af}$ = Higher heating value as received (ar), when dry or ash free (af)
$LHV_{dry/ar}$ = Lower heating value when dry or as received (ar)

All units above are in MJ kg^{-1}. The parameters of these equations are provided by the Phyllis Database (ECN, n.d.).

5.5. Summary: Data Collection

Four different types of data must be collected in order to perform a LCA for the case studies identified in **Chapter 4**. These include inventory data of inputs and outputs to the processes and data on how these may range due to variability, or how uncertain they may be. Data is also needed to convert the inputs into GHG emission factors, and, depending on the LCA methodology applied, data is needed on how GHG emissions are allocated between co-products. The following three chapters provide inventory data, variability and uncertainty data for wheat cultivation (**Chapter 5**), Miscanthus cultivation (**Chapter 6**) and 1st and 2nd generation bioethanol production (**Chapter 7**).

Chapter 6. The Wheat Model

This chapter describes the model dataset collected for the wheat grain and straw case study. The dataset aims to represent current wheat cultivation in the UK, including ranges in GHG emissions due to practices, geography, temporal aspects or due to uncertainty. **Figure 6-1** shows the system boundaries of the wheat cultivation system, highlighting the main sources of GHG emissions accounted for in this study.

The aim is to use the results from the case study as a basis for calculating the GHG emissions from wheat grain production for food and biofuels, and wheat straw for biofuel production. The case study will also be used to calculate the GHG emissions according to the rules presented by the GHG reporting methodologies.

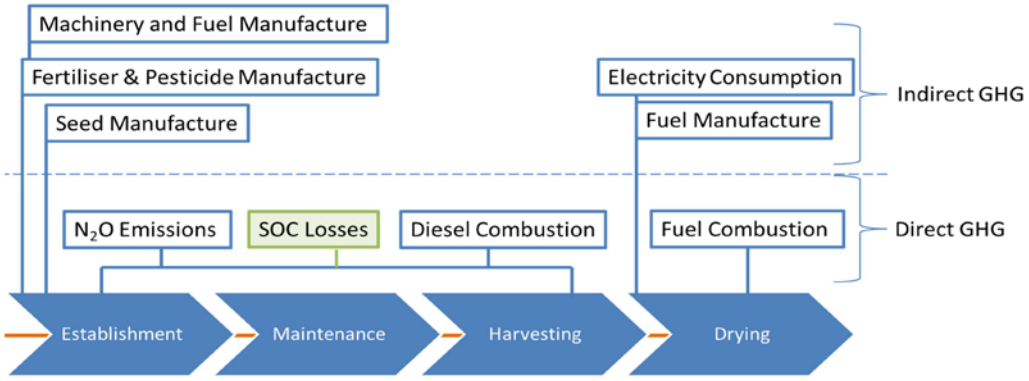

Figure 6-1. Summary of wheat cultivation system.

This chapter describes the process by which information regarding the inputs and outputs from the wheat cultivation process is collected within the system boundaries of the study. The inputs to wheat are first discussed, including fertilisers, pesticides and fuel. The yields from a wheat crop are discussed as well as sources of land-based GHG emissions, such as N_2O and carbon changes in soil.

6.1. Inputs to Wheat Cultivation

The following sections describe the main material inputs to the site required for cultivation of wheat and straw harvesting. Inputs are defined as materials that are delivered to the farm and consumed for the purposes of wheat cultivation (**Table 6-1**). These are identified through data collection from the farmers, growers' guides and from statistical and academic literature resources and are described in the following sub-sections.

Table 6-1. Inputs, outputs and sources of GHG emissions in wheat cultivation.

Inputs	Outputs
• Fertiliser	• Grain
• Pesticides	• Straw
• Seed	
• Fuel	
• Machinery	

6.1.1. Fertiliser Inputs

The fertiliser inputs to wheat is a particularly large issue, as there are various types of fertilisers that can be applied to a wheat crop. Also, as the fertilisers are applied differently across farms they must be examined separately. The issues are discussed in the following sub-sections:

- **Nitrogen-Based Fertilisers (as N)** – applied to all farms as ammonium nitrate, ammonium sulphate, urea or other forms
- **Potassium-Based Fertilisers (as K_2O)** – applied to some farms mainly as potash
- **Phosphorous-Based Fertilisers (as P_2O_5)**– applied to some farms mainly as triple-superphosphate
- **Organic fertilisers** – applied in various forms to some farms

The farmer questionnaires provided data on fertiliser use with varying levels of data quality. For example, farmers were given the option to provide a number for 'total nutrient applied', or alternatively provide information on quantities of specific fertilisers used. In 44% of questionnaires, farmers provided just the quantities of nutrients used, instead of specific details. Therefore, in order to represent a 'model hectare' of wheat some assumptions are made after consulting statistics and growers' guides.

6.1.1.1. Use of Nitrogen-Based Fertilisers

The yield of wheat is highly dependent on the application of nitrogen to the soil, and as it is highly mobile in soil, and is applied every year. To develop a 'model hectare' of wheat the following need to be identified:

- **The average, minimal and maximum application rates of N across farms**
- **The main forms in which N is applied in farms.**

Across farms in the sample the average N application rate is 319 kg N ha^{-1}, ranging between from 81 and 924 kg N ha^{-1}. Excluding that provided in organic form the average is 236 kg N ha^{-1} (81 to 415 kg N ha^{-1}). The recommended nitrogen application rates for winter wheat range between 160 and 250 kg ha^{-1}, depending on the intended end use of the grain (Agro Business Consultants Ltd, 2011; Nix, 2011). Deviations from the recommended rates may be caused by the nutrient content of the soil, or the Soil Nitrogen Supply Index (MAFF, 2000). Organic fertilisers may provide additional nitrogen, but may also provide other properties of soil amendment (Blanco-Canqui, 2012).

In the sample of 61 fields, the distribution of N application is positively skewed for artificial, organic and total N application (Z_{gi} = 2.341). It is therefore assumed that the collected data has a triangular distribution, with the 'best guess' and minimum and maximum figures provided based on the collected data. This is a suitable assumption considered the role of nitrogen in agriculture. The response to both yield and grain quality with fertiliser application has been studied from long-standing field trials performed by Rothamsted Research (Williams et al. 2006), and ADAS (Kindred et al., 2008b). The response curve is generally consistent, showing initial increasing responses to N fertiliser application, yet after a specified turning point, increased applications cause either a negligible increase or losses of yield.

Nitrogen fertiliser represents about 56% of the annual variable costs of a typical farm (Nix, 2011). One can assume that the results from the farmer questionnaires represent the economic optimum for bread wheat (**Figure 6-2**). A high variation of organic applications is seen in the results, as it may be applied for reasons other than nutrition.

In terms of the types of N-based fertilisers applied, the farmer data reports that on an average field, 54% of artificial nitrogen is applied in the form of ammonium nitrate, and 37% from urea. On average, about 21% of nitrogen is supplied in an organic form. UK statistics also report that the majority (51%) of fertilisers are applied as ammonium nitrate.

Data from literature and growers guides is used to modify the fertilisation rates from milling to feed wheat. Here an adjustment is made to the fertiliser requirements: feed wheat has a lower protein content than milling wheat (Smith et al. 2006). Milling wheat requires 40-60 kg ha^{-1} more N applications per hectare (Agro Business Consultants Ltd, 2011; MAFF, 2000; Nix, 2011a).

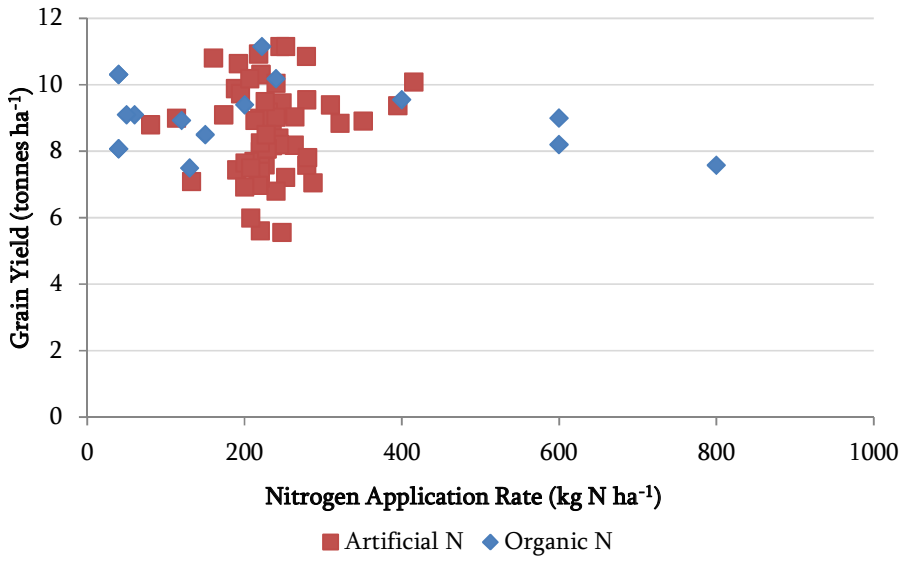

Figure 6-2. Fertiliser application rates in the farmer data.

6.1.1.2. Use of and Potassium-Based Fertiliser Requirements

Phosphorous and potassium can also be supplied from soil nutrient reserves or from fertiliser application. Applications of these nutrients are usually based on the planned rotation, rather than the single crop, as they do not easily leach from the soil to groundwater, so that applications can be quantified so that they benefit more than one rotation (Van Zeijts et al., 1999). It is therefore suggested by van Zeijts et al. (1999) that applications of P_2O_5 and K_2O are allocated between crops that they supply.

Within a single rotation, applications of P_2O_5 and K_2O should at least replace the nutrients that are removed from the soil and exported from the field within biomass. In the growers' guides (Agro Business Consultants Ltd, 2011; MAFF, 2000), P_2O_5 and K_2O application rates are specified according to if the straw is removed from the site or incorporated. Overall, **Table 6-2** summarises the fertiliser application rates assumed in this study.

Table 6-2. Summary of fertiliser application rates assumed in this study for milling wheat.

Fertiliser	Application Rate				
	Units per ha	Average	Proportion	Min	Max
Artificial N	kg N	235.5	100%	80.5	415.2
Urea		88.0	37%	30.1	155.2
Ammonium sulphate		20.1	9%	6.9	35.5
Ammonium nitrate		127.3	54%	43.5	224.5
Organic N	Kg	83.0	100%	0.0	509.0
Coffee Waste		63.4	76%	0.0	388.8
Farm Yard Manure		3.2	4%	0.0	19.5
Biosolids		0.6	1%	0.0	3.4
Compost		6.3	8%	0.0	38.9
Broiler litter		9.5	11%	0.0	58.3
Total P	kg P_2O_5				
Triple Superphosphate		17.3	100%	0	258.3
Total K	kg K_2O				
Potash		40.5	100%	0	280.0

6.1.1.3. Straw Removal Effects on Fertiliser Application

This section discusses how straw removal may cause nutrient loss in the soil. It is assumed that the nutrient off-take must be replaced with artificial fertilizers to avoid a drop in subsequent yields (Cherubini, 2010a; Punter et al., 2004). Replacement can be achieved by either artificial fertilisers or manure (Blanco-Canqui, 2012). When examining the farmer data, 40% ploughed in and 60% removed their straw. A Mann-Whitney U test showed there was no significant difference in N-based applications between fields that ploughed or removed their straw ($p=0.33$). Due to the sporadic nature of P_2O_5 and K_2O-based applications (many fields with zero application), there was not enough data to test for statistical significance.

There are various estimates for the value of the fertilizer penalty in the literature (**Table 6-3**). Therefore there is uncertainty in both the quantity and type of nutrients required to compensate for straw removal. Due to lack of data, a uniform distribution between minimum and maximum values is assumed. Residue incorporation is believed to be necessary for the recycling of nutrients, especially K_2O, P_2O_5, but less so for N, which is typically removed in the grain component (Whitbread et al., 2003). Some estimates of nutrient off-take are deduced from analyses of the nutrient content of straw (CORN, n.d.; NRCS, n.d.) or biophysical models (Gabrielle and Gagnaire, 2008). These may not represent nutrient availability in the field as the relatively high C:N ratios of cereal residues may lead to N immobilisation during decomposition and cause an increase in the N requirements for the following crop (Limon-Ortega et al., 2008). There can also be variation in nutrient contents of straws between sites and across years (Withers, 1991).

Table 6-3. Estimates of nutrient off-take in straw.

Resource	Nutrient Content (kg tonne straw^{-1})		
	N	P_2O_5	K_2O
Punter et al. 2004	19.4	3.4	33.7
Cherubini & Ulgiati, 2010	3.0[a]	2.8	2.2
Plant Nutrient Content Database (n.d.)	7.6	0.8	14.7
CORN (n.d.)	5.0	1.4	9.1
Tarkalson et al. 2009	7.4	1.1	9.4
Hollinger n.d.	5.5	1.7	13.6
Copeland & Turley 2008	5.0	1.3	9.3
Potash Development Association (n.d.)	-	1.2	9.5
Withers 1991		1.4	9.3
Brander et al., 2009b[b]			
HGCA 2009	-	1.2	9.5
Agro Business Consultants Ltd 2011 (low)	-	-	7.3
Agro Business Consultants Ltd 2011(high)	-	-	16.0
Gabrielle & Gagnaire,2008	-	-	-
Slade et al. 2009[b]	-	-	-
Average[c]	9.0	1.8	11.9
Minimum	0.0	0.8	2.3
Maximum	20.0	4.0	33.7

a. *Value is mid-point between low and high estimate.*

b. *Resource provided only emission estimates for the fertiliser requirements*

c. *Average does not include zeros*

6.1.2. Pesticide Requirements

Pesticides are applied to control fungal diseases, insect pests, and weeds, all of which are a threat to the overall yield of the crop. Data on pesticides use was not recorded in the questionnaires because the authors assumed they have a negligible contribution to the overall GHG balance of a crop, and it is suggested that the impact of not using them has a higher GHG impact than applying them (Berry et al. 2008). They are included in this study for completeness, and the Pesticide Usage Survey is used to estimate pesticide use on a generic wheat crop.

Pesticides include fungicides, insecticides, and herbicides and may include growth regulators too. Fungicides are generally applied for controlling general build-up of disease, though some are applied to tackle specific problems (Garthwaite et al., 2010). Out of all pesticides, herbicides are applied in the greatest quantities. Herbicides are applied through spray application, though it is also common to apply spot applications to problem areas (UK Agriculture, 2012b). Growth regulators are required to prevent lodging by restricting plant growth: either through gibberellic acid inhibition, or ethylene production (Edwards and Dodgson, 2005).

Table 6-4 summarises the pesticide usage in 2010 in the UK according to statistics. The year 2010 is selected so that the results of Webbs et al. (2010) can be cross-referenced. Due to lack of data, pesticide use is assumed to have a uniform distribution. Temporal variation is not included due to a lack of data. A sensitivity analysis will determine the impact of pesticides on the overall GHG emissions.

Table 6-4. Summary of pesticide usage on a typical wheat crop.

Source	Pesticide	Total Applied to Cropland (tonnes)	Application Rate (kg ha^{-1})*
FERA Statistics	Fungicides	2,611.4	1.35
	Herbicides	2,455.8	1.27
	Insecticides	92.7	0.05
	Growth Regulators	2,148.4	1.11
	Total	**7308.3**	**3.77**
Webbs et al. 2010	**Total pesticide**		**4.55 (3.43-5.15)**

** Based on a total cropping area of 1,939,000 in 2010* (DEFRA et al., 2010).

6.1.3. Seed Requirements

Seed is usually deemed an 'insignificant' component of the LCA (Bauen et al., 2008), and therefore this is not researched in great detail here.

Table 6-5 lists typical application rates of seed per hectare, based on information provided in the farmer data and growers' guides.

Table 6-5. Seed application rates.

Source	Seed Application (kg Seed ha^{-1})		
	Average	Min	Max
Growers' Guides	166	150	200
Farmer Data	151	90	200

6.1.4. Fuel Requirements for Cultivation

This section details the diesel fuel consumption rates for wheat cultivation and straw baling. Diesel is used for all mechanical operations carried out by tractors. In any crop, cultivation operations may vary in intensity between sites. Farmers will manage their soils according to their soil type and rotation. The following section is structured as:

- **Operations performed** – identifies the operations performed on an 'average' wheat crop.
- **Diesel fuel consumption rates –** estimates of diesel fuel consumption rates.

The reason that diesel fuel must be estimated is because no farmers provided fuel consumption rates for specific fuel operations. All farmers provided were a list of operations, maybe because this requires very little collection effort. There are no relevant statistical data on diesel fuel requirements for cereal cultivation.

6.1.4.1. Operations Performed

In this study, the operations for an 'average field' were calculated based on the average times each operation was performed across the sample. The results are shown in **Table 6-6**.

Table 6-6. Operations involved in wheat cultivation on an 'average' farm (collected from farmer questionnaires).

Stage	Operation	Number of Times Performed		
		Mean	Min	Max
Establishment	Heavy Ploughing	0.3	0.0	1.0
	Power Harrowing	0.1	0.1	1.0
	Subsoiling	0.1	0.0	1.0
	Heavy Discing	1.0	0.0	3.0
	Light Discing	0.1	0.1	2.0
	Drilling	0.9	0.9	1.0
	Drilling (Combined)	0.1	0.1	1.0
	Rolling	1.0	1.0	2.0
Maintenance	Spraying	6.6	0.0	14.0
	Fertilising	3.8	1.0	5.0
	Muck Spreading	0.3	0.0	1.0
Harvesting	Combine Harvesting	1.0	1.0	1.0
	Tractor and Trailer (14t)	1.0	1.0	1.0

6.1.4.2. Estimates of Diesel Fuel Consumption

In the literature, there are various estimates of diesel fuel consumption rates for various farm operations. These are reviewed and the ranges of fuel consumption rates are assessed. It is also possible to estimate fuel consumption rates using the following equation:

Equation 6-1. Estimating diesel fuel consumption from a farm operation (Matthews et al., 1994).

$$KWh_{fuel} = \left(\frac{KW \times t}{e}\right) PTO$$

Where:

KWh_{fuel} = Fuel consumed in (KWh)
KW = Power rating of engine (KW)
t = Time required to perform operation (hr)
e = Engine efficiency (assumed at 0.25 litres hp hr^{-1})
PTO = Power take off, percentage of total engine power required for operation (%)

As the equation is multiplicative, the result is equally sensitive to each parameter. To convert KWh_{fuel} to litres ha^{-1}, a conversion rate of 10.5 KWh $litre^{-1}$, or 37.9 MJ $litre^{-1}$ is used and the conversion rate between horsepower and KW is 0.75 KW per hp.

Horse power and work rate data for specific operations are provided in growers' guides. To test the validity of the equation, the estimated diesel consumption rates were compared with the recommended horsepower (hp) for specific operations from the Nix and ABC handbooks. Nix (2011) specifies that very heavy work, such as subsoiling or deep ploughing is carried out by 165 hp tractors. Heavy farm work, like ploughing or drilling is carried out by 130 hp tractors, and 110 hp will suffice for all other work. The ABC farm guide recommends larger tractors, between 260-310 hp for very heavy work, 180-230 hp for heavy work and 150-180 hp for others work (Agro Business Consultants Ltd, 2011). Work rates also differ between the two resources.

The diesel fuel consumption rates were estimated according to **Equation 6-1**. The results were compared with data from literature and the results are shown in **Figure 6-3**. A total of 9 literature sources were consulted for 9 operations. The majority of the data points from literature originate from Lal (2004a), which provides an average, low and high estimate of fuel consumption rates in the agricultural sector. Lal fails to correlate the average fuel consumption rates with an average hp rating.

Equation 6-1 estimates a higher fuel consumption rate for subsoiling, heavy discing, harrowing and ploughing than that reported in the literature. These are operations which involve a great deal of interaction with the soil. There is also a larger range in fuel consumption estimates in these operations, mainly because there are various levels of tillage that can be adopted. For

example a chisel plough will cause minimal soil inversion, and a mouldboard plough is more intensive and requires a greater amount of energy to overcome the friction between the implement and the soil; both are however, forms of ploughing. Also, within the same plough type, soil factors can affect the required fuel consumption to carry out an operation (Williams et al. 2006). Tractor performance for a given operation will depend on the implement type and shape (Lal, 2004b), and also on soil physical properties such as bulk density and water content, which ranges between sites and even within the same field (Perfect et al., 1997).

There is very little variation in operations that require passage over the soil, such as spraying or broadcasting and planting (**Figure 6-3**). Cereal harvesting, is an exception here, as the theoretical estimate is lower than found in the literature. Harvesting can be affected by whether or not the straw is chopped, which is not always stated in the literature resources.

For the purposes of this study, the two datasets are combined to cover the highest and lowest possible estimates for farm machinery fuel consumption. As there is insufficient data to determine the probability density function of the distribution of fuel consumption rate, a uniform distribution between minimum and maximum likelihoods is assumed. The sensitivity of the results to diesel fuel consumption will be tested. The fuel consumption rates are listed in (**Table 6-7**).

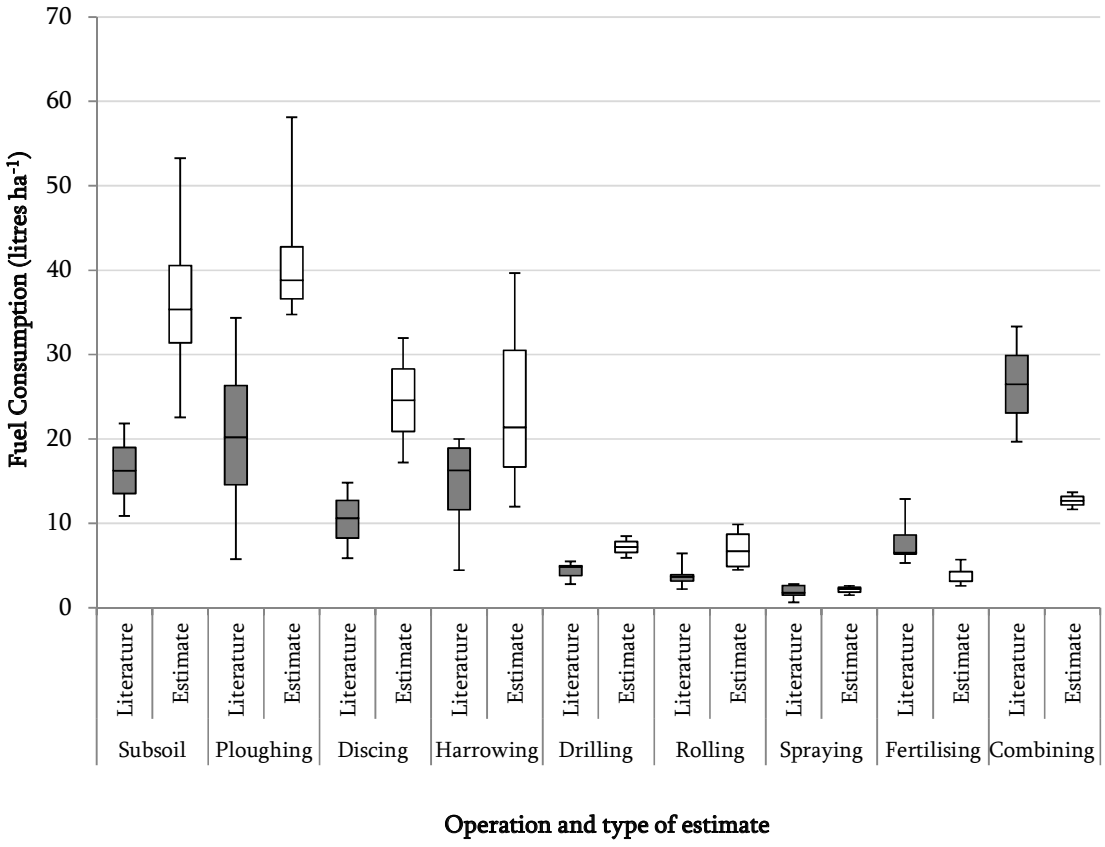

Figure 6-3. Fuel consumption rate of typical farm operations based on data collected from literature and estimated based on theoretical data.

Table 6-7. Estimates for fuel consumption used in this study.

Operation	Fuel Consumption Rate (litres ha^{-1})		
	Average	Minimum	Maximum
Heavy Ploughing	27.7	5.8	58.1
Power Harrowing	18.6	4.4	39.7
Subsoiling	26.5	10.9	53.3
Discing	16.1	5.9	32
Drilling	5.4	2.8	8.5
Rolling	5.2	2.2	9.9
Spraying	1.9	0.6	2.8
Fertilising	6.2	2.6	12.9
Combine Harvesting	25.3	11.7	33.3
Baling	12.5	2	30.8

6.1.5. Fuel Requirements for Straw Baling

The diesel fuel consumption for straw baling and harvesting is estimated after reviewing both published resources and farmers guides (**Table 6-8**). Where the fuel consumption is estimated it is based on an equation provided by Matthews et al. (1994), and assumes a tractor horsepower of 200, and PTO of 0.85 for harvesting.

Table 6-8. Estimates for Fuel requirements for straw baling.

Resource	Fuel Consumption (Litre tonne $straw^{-1}$)	GHG Emission (kg CO_2 eq. tonne $straw^{-1}$)
Nemecek	8.6	23.1
Pers. Com. Contractor	3.5	9.3
Lal, 2004b	0.6	1.6
Lal, 2004b	1.2	3.2
Lal, 2004b	1.8	4.9
Williams et al., 2006	1.7	4.6
Bullard and Metcalf	7.7	20.7
BEAT	8.8	23.6
Nix, 2012*	3.1	8.2
Min	**0.6**	**1.6**
Max	**8.8**	**23.6**

6.1.6. Fuel Requirements for Grain Drying

Generally, grain drying is recommended to reduce the moisture content of grain to 14% or lower in order to prepare it for storage (HGCA, 1998). In the farmer questionnaires only the method of drying and moisture content before and after drying was recorded. The fuel consumption for drying was calculated using BEAT, which includes bulk, batch, continuous and natural drying (**Table 6-9**).

In the sample of farms, 70% of farmers dried their grain. The majority (79%) utilised a diesel-based drying system to dry their grain either continuously (58%), via bulk (14%) or batch (7%), with few others using electricity or LPG. Energy requirements for drying were derived from BEATv2, which estimates the energy input per tonne water evaporated during drying (AEA Technology and North Energy Associates, 2010).

Table 6-9. Energy requirements for evaporation of one tonne of water (twe).

Drying Method	Energy Requirement (MJ twe^{-1})
Batch	3959
Bulk	2074
Continuous Cooling	3083
None/natural	0

6.1.7. Use of Farm Machinery

Assuming one tractor is used to run the implements, plus combined harvesting, baling, a total of 3 machines and 11 implements are assumed to be required to cultivate a typical hectare of wheat. **Chapter 4** identifies two main ways of estimating GHG emissions from machinery manufacture: by examining the relative material composition of the machine and applying emission multipliers for each material. As the cost multipliers by North Energy Associates et al. (2007) have not been updated since 2004, it was decided to use emission factors in the Ecoinvent Database for machinery construction. These are based on the composition of 1 kg of the machine. The weights of the machines and implements are taken from Williams et al. (2006). The results are provided in **Appendix 3**. The total manufacture of the machinery is allocated to the time required to perform the given task set out to provide the functional unit of the study. It is assumed that the machinery and equipment has a total potential lifetime of 7000 hours (EcoInvent, 2007).

6.1.8. Summary of Inputs

Inputs to wheat are mainly variable due to practice though there is some uncertainty in the type of wheat that is being grown and there is a general lack of statistics of the types of wheat being grown in the UK. For example, a default value of "wheat" could refer to milling or feed wheat. There is also winter and spring wheat however this difference is not examined here. The yields of spring wheat are usually much lower than winter wheat, for both grain and straw (Nix, 2011a). Between these types there are small differences in the yields and fertiliser requirements. If wheat is intended to be used for bioethanol, it is likely to be feed wheat, whereas if intended for food then it may be milling wheat. Therefore, knowledge of the final intended use can reduce some of the uncertainty due to variety.

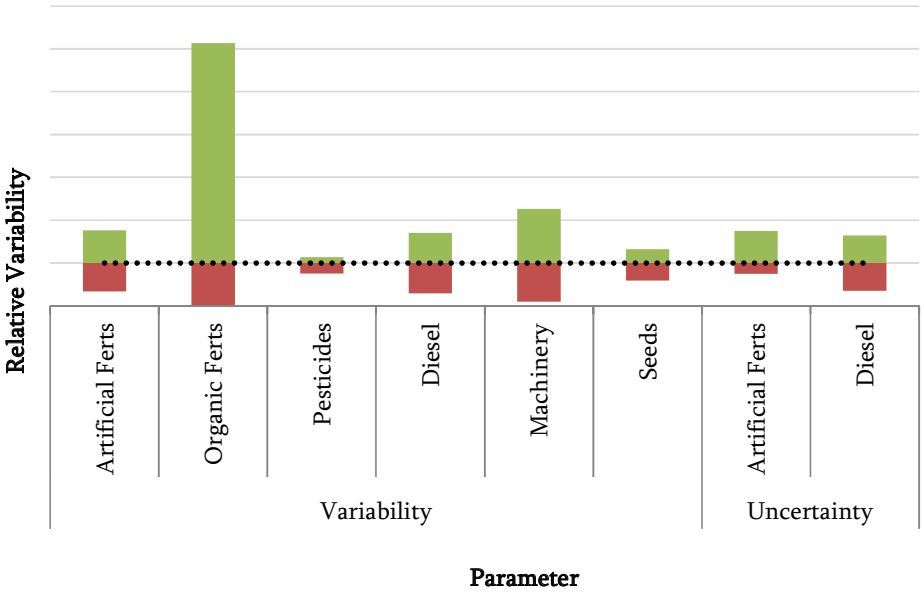

Figure 6-4. Relative variability and uncertainty of inputs to wheat cultivation.

Figure 6-4 shows the percentage variation from the average application rate in each of the site inputs. It shows that organic fertiliser is the most variable and uncertain input, mainly because farmers applied between 0 and 509 kg N ha^{-1} on their sites. Artificial fertilisers, diesel and machinery all vary to the same extent. There is some uncertainty in artificial fertiliser application due to the straw removal penalty and variety of wheat used. Seed and pesticides rates are more consistent.

6.2. Land Use Change Effects

This section discusses the GHG emissions that may occur due to direct land use change (DLUC) in wheat. DLUC can occur due to conversion of land into arable land or changes in residue management. The calculation methodology presented in the RED and the supporting documentation (EC, 2010b, 2009a) is used to calculate LUC. In the calculations the UK is defined as "Cool Temperate Moist" (Bickel et al., 2006), based on a mean annual temperature of 8.8 °C and mean annual precipitation of 1331mm (Met Office, 2012a).

6.2.1. DLUC Due to Conversion to Arable Land

DLUC is introduced in **Chapter 2**. It involves a direct change of land use from one type to another. For example, forestland could be converted to arable land, and the direct consequence of this would be the loss of organic SOC in the soil and biomass in the forest stand (Hillier et al., 2009). The IPCC and RED calculation methodology will include such a change in land use if it occurred within 20 years of the point of accounting. Although DLUC may not be relevant to the farms studied here, it has been identified as a significant source of GHG emissions in agriculture (Roches et al., 2010; Röös et al., 2010), hence must be examined here.

The total carbon content of a site is contained within the soil and the biomass. **Table 6-10** contains the carbon content of forestland, grassland and arable land, according to the calculations laid out in the supporting documentation to the RE, which follows the methodology provided in the IPCC. The minimum and maximum potential changes in total carbon when forest and grassland are converted to arable land are provided. It is assumed that the total change is averaged out over a period of 20 years.

Table 6-10. Total carbon stock and losses of carbon when forest and grassland are converted to arable land, calculated using the IPCC and RED calculation methodology.

Land Type	Carbon Content (tonnes CO_2 eq. ha^{-1})			Change with Conversion to arable land (tonnes CO_2 eq. ha^{-1} $year^{-1}$)	
	Parameter	Min	Max	Min	Max
Forest Land	Soil	260.3	130		
	Biomass	51.3	289.7	2.2	28.3
	Total	311.7	766.3		
Grassland	Soil	329.4	603.2		
	Biomass	11.4	29.7	3.7	21.7
	Total	340.8	632.9		

6.2.2. Changes in Residue Management

Very few studies include the effects that straw removal has on changes in SOC. **Therefore this study presents a novel account of the potential significance that has on the specific GHG emission saving targets of straw-based bioethanol.**

Changes in SOC may occur due to straw removal. Estimates in the literature for such changes show considerable range (**Table 6-10**) and there is uncertainty in what the estimates refer to, for example they do not state whether can be achieved alongside minimum or conventional tillage.

Table 6-11. Literature estimates of SOC changes due to straw removal or incorporation.

References	Carbon Change (tonnes CO_2 eq. ha^{-1} $year^{-1}$)	Timescale Applicable (years)
Smith et al. 2012, (low)	0.39	10
Smith et al. 2012, (high)	0.78	10
Gabrielle & Gagnaire 2008	2.35	1
Bickel et al. (2006)	0.26	1-20
Average	**0.95**	
Min	**0.26**	
Max	**2.35**	

6.3. Outputs from Wheat Cultivation

This section describes the yields of **wheat grain and wheat straw,** which are the two outputs from the case study.

6.3.1. Wheat Grain Yields

Applications of N are the single most important factor affecting the final yield and quality of cereal grain, therefore some assumptions must be made when considering yield. The following sections examine wheat yields according to:

- **Temporal variation over time**
- **Spatial variation due to location.**

6.3.1.1. Temporal Variation in Yields

UK Statistics show that grain yields have increased over the last three decades (Farming Statistics, 2012), with a noticeable advance in the 70's (**Figure 6-5**) after the introduction of straw shortening compounds. These reduce the height of straw so that less crop is lost due to lodging (M. Carver, pers. com. 2011). Since then breeding and intensive production have led to yield increases until the late 90's where it appears to have stabilised around an average yield of 7.8 tonnes ha^{-1}, with a standard deviation of 0.3 tonnes ha^{-1} (Farming Statistics, 2012). More recent dips in yield in 2001, 2007 and 2012 are mainly due to adverse weather conditions (DEFRA et al., 2009; Yan and Boies, 2013).

There is some uncertainty in the statistics, as there is a lack of records of yields of specific varieties or types of wheat. It is difficult to find data on the coverage of milling and feed wheat. According to growers' guides milling wheat has a 7-8% lower yield than feed wheat (Agro Business Consultants Ltd, 2011; Nix, 2011a). The yields of milling and feed wheat are listed in **Table 6-12**, and are compared to the 10 year average.

In terms of future yields, linear regression is one method of prediction of wheat grain yield. The temporal data most closely matches a polynomial regression (R^2= 0.9532). The method predicts a yield increase to 9.26 tonnes ha^{-1} by 2020, which is also predicted by a model developed by Ewert et al. (2004). It has been suggested that future yields may be restricted due to UK limitations on N fertiliser use (T. Dawkins pers. com. 2013).

Table 6-12. Temporal variation in yields for winter wheat (units is tonnes ha^{-1} $year^{-1}$).

Crop	Feed Wheat		Milling Wheat		Average	
	Average	S.D	Average	S.D	Average	S.D
2010	8.9	1.4	8.2	1.3	8.6	1.8
10 Year Average	-	-	-	-	7.8	0.3

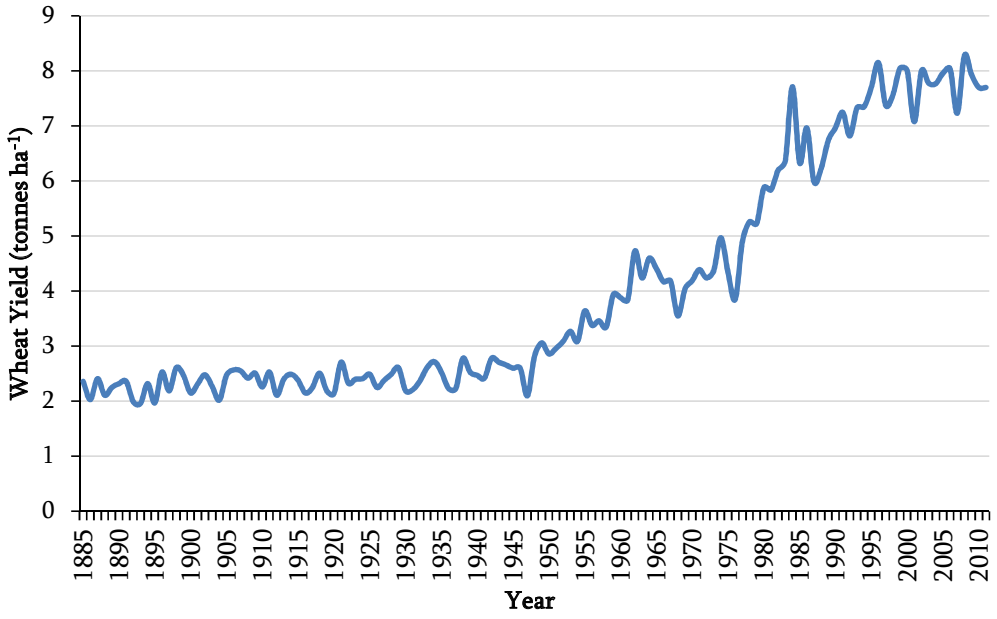

Figure 6-5. Yields of wheat over time (Farming Statistics, 2012).

6.3.1.2. Spatial Variation in Yields

The national statistics of wheat yields do not provide data on spatial variation. A region-specific analysis by Webbs et al. (2010) showed there were no statistical differences between regions in the UK (Yan and Boies, 2013). An average yield for winter wheat was estimated at 6.6 tonnes ha^{-1}, (Webbs et al., 2010), which is lower than that reported in both national statistics and that reported by the farmer questionnaires (8.3 tonnes ha^{-1}). The differences may be due to the varieties grown, for example Webbs et al., 2010 only examined winter wheat.

Following the methodology in **Appendix 7**, a statistical test of skewness and kurtosis indicates that the grain yields in the farmer data are not significantly skewed (Z_{gi}= 0.042), and a Pearson's Chi Squared test indicates that the distribution of the grain yield is not significantly different to a normal distribution (X^2 = 113, d.f = 120, bin size = 0.4). Therefore it is assumed that the grain yield is normally distributed; a realistic assumption when considering typical patterns observed in Mendelian genetics (Falconer 1996).

6.3.2. Wheat Straw Yields

This section discusses wheat straw yields, of which there is limited data. Straw yields are not reported in UK statistics. Typical straw yields are contained in growers' guides, with yields falling between 3 and 5 tonnes ha^{-1} (Agro Business Consultants Ltd, 2011; Nix, 2011a). Straw yields are not as accurately reported as grain. They tend to be recorded to the nearest half tonne as Heston straw bales have an average weight of 500 kg, and therefore the straw yield is estimated based on a yield of bales per hectare.

The farmer data recorded straw yields for all but 2 sites. The average yield of straw was 3.6 tonnes ha^{-1}, ranging from 2.5 to 5.5 tonnes ha^{-1}. The data suggest that the yield of straw is positively skewed (Z_{gi}= 3.424, See **Appendix 7**). Therefore it is assumed that the straw yield has a triangular distribution (**Figure 6-6**).

It is assumed that between 66% and 75% of cut straw can be removed from the site by modern harvesting technology (Perlack et al., 2005; Webbs et al., 2010). It is possible that improvements in agricultural technology could be made to reduce the gap between potential and actual straw yields (Ewert et al., 2004). Farmers will only modify straw harvesting so long as it does not reduce the efficiency in which they collect grain (Personal communication: C. Dawson, 2007). Another option to maximise straw availability is to reduce the use of straw shorteners, but this leaves the crop at risk of wind damage or lodging. It is estimated that losses up to 20% are experienced on the field during harvesting and pressing, and 10% is lost during transport and storage (Labalette & Panoutsou, 2006).

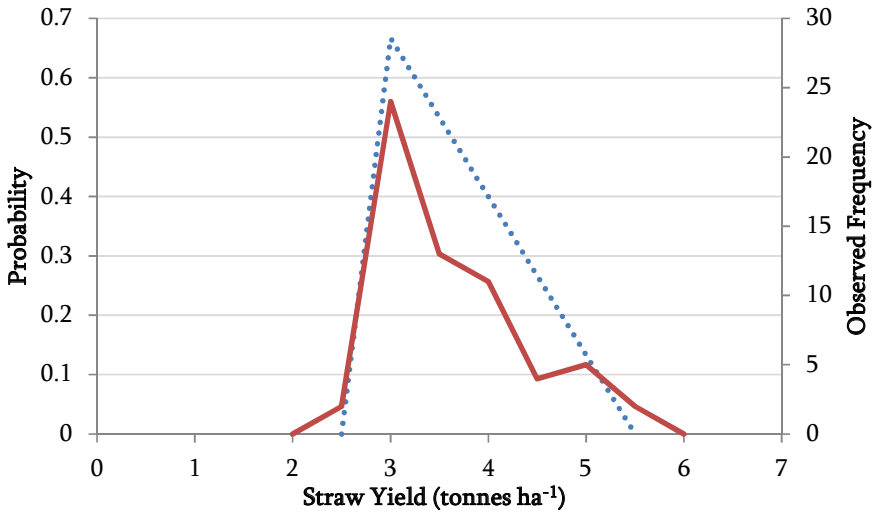

Figure 6-6. Fitting a triangular distribution to wheat straw yields.

6.4. The Economic Value of Wheat and Straw

The prices of both wheat grain and straw have been recorded in detail since 2000. Prices of wheat grain are collected four times a month, and once a month for straw (Gov.UK, 2013). By combining the data and looking at the 5 year rolling averages for wheat and straw (**Figure 6-7**), it is clear that prices of both have doubled over the last 12 years.

Unlike yield data, statistics are reported for both milling and feed wheat, the latter being between 10 and 16% lower in price. Despite the steady increases of price, it is expected that global wheat prices will level out between £160-180 $tonne^{-1}$ between 2014 and 2020 (OECD et al., 2011).

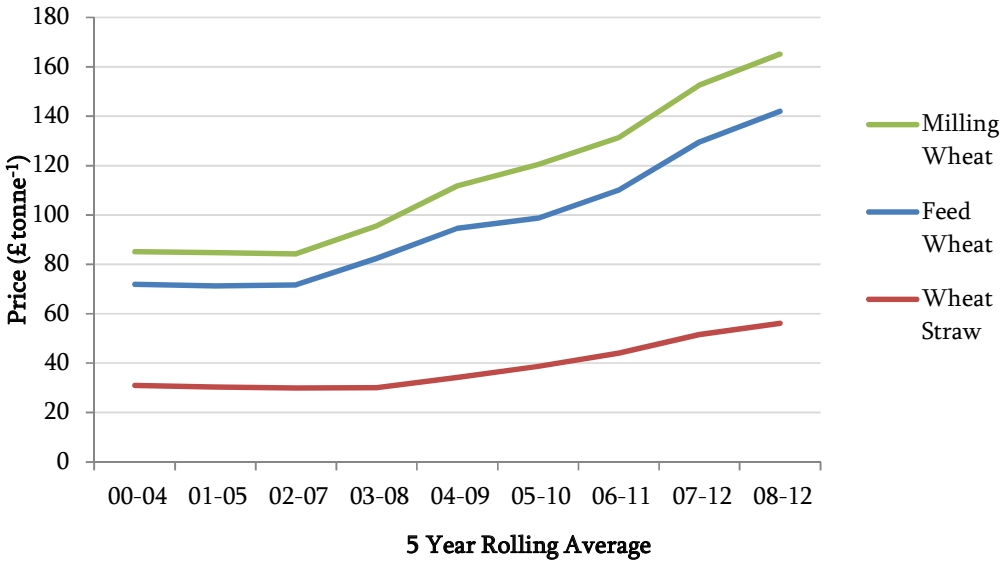

Figure 6-7. Five-year rolling average of prices of milling wheat, feed wheat and wheat straw.

The prices recorded in National statistics are based on grain that is ready for wholesale price and baled straw, i.e. they are 'ex-field' prices. Some processing has occurred between the field and the point at which the price is determined. In the field, the two components: grain and straw, are separated in the combined harvester. After combine harvesting straw is left on the field, and a baler must pass over the field in order to collect it and create a "handle-able" product. There is no price information available on grain or straw at this point, so a 'shadow price' must be determined to represent the 'in field price'. Shadow prices are defined in **Chapter 2**, **Section 2.5**. A shadow price is determined by deducting the price of grain drying and storage from the wholesale price of the grain. Likewise, the shadow price of straw is determined by deducting the price of baling.

Drying and baling costs were collected for the last 12 years from each year's editions of the Farm Management Pocket Books and the results are shown in **Figure 6-8**. There is much variation in the relative prices of wheat grain and straw, though over the last 10 years the average allocation to feed wheat is 89.3%, and straw 10.7%. The maximum and minimum allocation to feed is 94.9% and 81%. Due to the higher value of milling wheat, the average allocation is 91.1%.

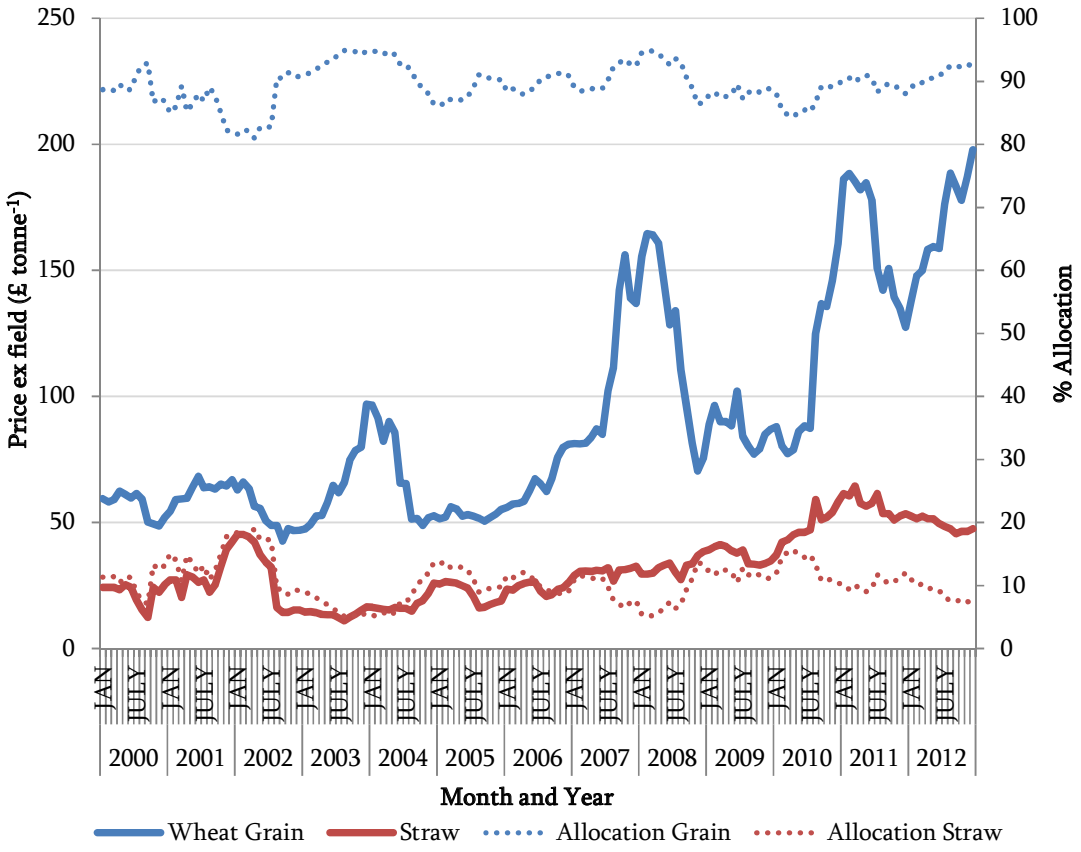

Figure 6-8 Prices and allocation of wheat grain and straw over time.

6.5. The Energy Content of Wheat Grain and Straw

As described in **Chapter 4**, the energy content of solid biomass is calculated following the Milne equation, using data from the Phyllis Database (ECN, n.d.). The calculated energy content of wheat grain and wheat straw are provided in **Table 6-13**.

Table 6-13. Energy contents of wheat grain and wheat straw assumed in this study.

Biomass	M. C	Hydrogen	Ash	HHV_{daf}	HHV_{ar}	HHV_{dry}	LHV_{dry}	LHV_{ar}
	%			MJ kg^{-1}				
Wheat Grain	20	5.1	7.3	19.6	14.5	18.2	17.1	13.2
	14				15.6	18.2	17.1	14.3
Wheat Straw	20	5.5	5.0	19.4	14.7	18.4	17.2	13.3
	15				15.6	18.4	17.2	14.3

6.6. Substitution Credits

It is generally accepted that there are no substitution credits for wheat straw as it does not displace any land-grown products (Bauen et al., 2010). Often in LCA studies it is either allocated environmental impacts or ignored. Though it has been suggested that Miscanthus and other products could be used to substitute for straw (Brander et al. 2009b); it not assumed that straw substitutes the production of Miscanthus and other products.

6.7. Summary: One Hectare of Wheat

Figure 6-9 provides a summary of the total economic value and energy content from one hectare of feed and milling wheat in the UK. The results show that milling wheat has a higher net economic value compared to feed wheat, despite a 7-8% lower yield. When examining the energy content on a per-hectare basis milling wheat has a lower net energy content. Here, it can be seen that straw has a higher 'output' per hectare when allocated by energy content because, on a per-LHV basis, wheat straw has a higher relative net energy-content to grain than it does in price. As wheat grain and wheat straw have similar energy contents (between 13 and 14 GJ tonne^{-1}), allocating by mass will give similar results to energy content.

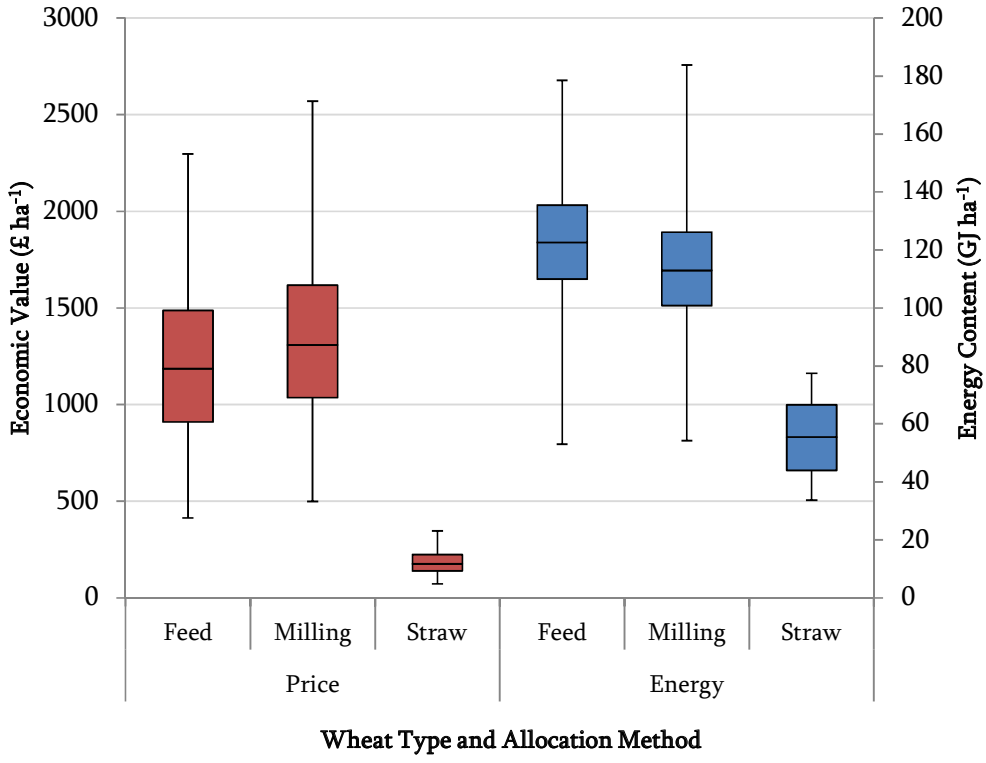

Figure 6-9. Relative economic values (red) and energy contents (blue) of feed and milling wheat and straw.

6.8. Summary: Wheat Cultivation

Wheat is an annual cereal crop that requires a certain level of management and inputs. Such inputs are necessary to ensure yields are achieved as expected, as well as the quality of grain being suitable for the intended final end-use. In this study, it has become apparent that there is some range in the overall inputs to the crop, especially regarding the use of organic fertilisers. The relative importance of these inputs will be determined when the impact assessment is performed in **Chapter 9**.

There is also uncertainty with the effects of straw removal on the GHG emissions from the fertiliser penalty, and the effect of SOC change in soil. Wheat straw can either be incorporated or baled. When incorporated, it is expected that straw undergoes decomposition, and in the process returns nutrients to the soil and a component of the biomass is stored in the long-term carbon storage pool as SOC. When straw is removed from soil it is assumed that these processes do not occur and there is a loss of SOC and replacement nutrients are required.

In summary, in wheat cultivation there is a combination of uncertainty and variations in:

- Yield of wheat and straw
- Fertiliser application rates
- Fertiliser types applied
- Fertiliser penalties from straw removal
- SOC losses due to straw removal

The net output of wheat can be examined in a number of ways: as pure masses of product, of the economic value of products, and the energy content of the products. Each option sees a different 'net output' per hectare and will affect how GHG emissions from cultivation are allocated between wheat grain and straw. It is assumed that no substitution credits are relevant to straw.

The following chapter examines the cultivation of Miscanthus.

Chapter 7. The Miscanthus Model

This chapter describes the model dataset collected for the Miscanthus case study. **Chapter 5** describes how the data is collected from growers and contractors. **Therefore a novel study of commercial-scale Miscanthus cultivation is presented here.** This is compared with data from literature.

This study also provides an in-depth account of the processes involved in commercial-scale rhizome cultivation: which is rarely included in studies. **This study also presents a novel account of the GHG implications of 'green' harvesting Miscanthus,** i.e. before it has entered the senescent winter phase. The benefits of doing so are increased yields, which are discussed in **Section 7.4**.

The aim is to the case study as a basis for calculating the GHG emissions from Miscanthus for use in 2nd generation bioethanol production. **Figure 7-1** shows the system boundaries of the Miscanthus cultivation system, highlighting the main sources of GHG emissions accounted for in this study. The system boundaries of the rhizome cultivation study are shown and discussed in **Chapter 4**.

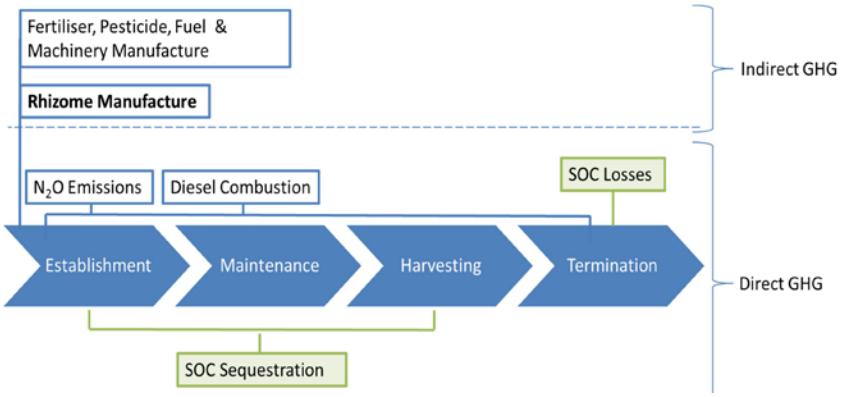

Figure 7-1. System boundaries of the Miscanthus cultivation system.

This chapter follows the same structure as that set out in the wheat case study in **Chapter 6**: describing the inputs and outputs from the Miscanthus cultivation process collected within the system boundaries of this study. The yields of rhizomes and Miscanthus biomass are also discussed as are the impacts of LUC when Miscanthus is planted on arable, grassland or forest land.

7.1. Inputs to Miscanthus Cultivation

The following sections describe the main inputs to the site required for cultivation of Miscanthus (**Table 7-1**). These are identified through data collection from the contractors and from literature resources.

Table 7-1. Inputs, outputs and sources of GHG emissions in wheat cultivation.

Inputs	Outputs
• Rhizomes	• Rhizomes
• Fertiliser	• Miscanthus
• Pesticides	
• Fuel	
• Machinery	

7.1.1. Rhizomes

In the literature, planting densities of Miscanthus crops range between 12,000 (Atkinson, 2009) and 20,000 (DEFRA, 2007) rhizomes ha^{-1}. For a rhizome multiplication crop, no information is available in the literature. Contractors reported that a planting density of 40,000 rhizomes ha^{-1} is used to ensure a high rhizome yield (I. Webber pers. com. 2012).

A sample of 27 rhizomes saw that the majority weigh up to 20 grams (Newman, 2003). The contractor estimated that 1 tonne of rhizomes represents approximately 20,000 rhizomes, or a weight of 50 g $rhizome^{-1}$ (M. Carver pers. com. 2012). Therefore a minimum and maximum rhizome weight of 20 and 50 grams represents a total rhizome mass of 0.24-1 tonnes ha^{-1} for a main Miscanthus crop and 0.8-2 tonnes ha^{-1} for a rhizome crop.

7.1.2. Fertiliser Requirements

The fertiliser requirements for Miscanthus is a particularly large issue, as there is no currently no consensus on the nutrient requirements of the grass or its yield response to fertilisers (Smeets et al., 2009). The following section is structured as:

- **Estimates of fertiliser requirements in Miscanthus** – reviews current knowledge.
- **Yield responses to fertiliser application** – reviews studies on yield responses to fertilisers.
- **Nutrient use efficiency and cycling in Miscanthus** – reviews evidence of translocation of nutrients within the Miscanthus plant.
- **Harvesting time and nutrient demands** – to deduce the nutrient demands of harvesting in autumn and spring.

7.1.2.1. Fertiliser Requirements in Miscanthus

As only a small number of studies have been conducted on nutrient requirements of Miscanthus, the exact fertiliser requirements are not yet defined (DEFRA, 2010). The commercial Miscanthus growers claimed they had never applied N to their sites, but would apply a top dress of 40 kg K_2O ha^{-1} in year 10 (T. Barton pers. com. 2012).

Some studies recommend single fertiliser applications at establishment, whereas some suggest yearly or variable applications (**Table 7-2**). The DEFRA guidelines suggest applications of 150 kg N ha^{-1} are applied in the first two seasons in organic form (DEFRA 2007; Hall 2003).

In contrast, the Fertiliser Manual of 2010 (DEFRA, 2010) recommends that very little N is applied in the first two years as this encourages weed growth and is more likely to run off. The Fertiliser Manual of 2010 (DEFRA, 2010) recommends annual applications of 60-80 kg N ha^{-1}, in organic form, are then recommended after years 2-3.

Literature demonstrates higher K_2O demands by Miscanthus than for P_2O_5 fertilisers (Christian et al., 2008), and Miscanthus biomass is said to have a relatively high K_2O content (Himken et al., 1997). The contractors reported only applying a single application of 40 kg K_2O as potassium chloride after a period of 5-10 years. Applications of K_2O were made in 8 out of 14 years to a Miscanthus crop in the UK, and P_2O_5 only twice during this period (Christian et al., 2008).

Table 7-2. Fertiliser application rates for Miscanthus in the literature.

Application Frequency	Application Rates in the literature (kg ha^{-1} yr^{-1})			References
	N	P_2O_5	K_2O	
Establishment Only	0 - 130	13 - 130	67 – 300	(Bullard & Metcalf 2001; DEFRA 2010; Fazio & Monti 2011; Himken et al. 1997; Monti et al. 2009; Styles et al. 2008; Styles & Jones 2007)
Yearly	0 -100	9 - 50	89 – 200	(Ercoli et al., 1999; Felten et al., 2013; Lewandowski et al., 1995; Smeets et al., 2009)
Variable		6 - 36	72 – 100	(Clifton-Brown et al., 2007; Gilbert et al., 2011)
Contractor	0	0	40	Applied at least every five years, not annually.

7.1.2.2. Yield Responses to Fertiliser Application

Applications to Miscanthus rely on identifying the crop's nutrient requirements so that excess additions are not made (Smith & Slater 2010). The idea that Miscanthus has minimal nutrient requirements stems from field trials where there is clearly no yield response to applied N is demonstrated (Christian et al., 2008; Danalatos et al., 2007; DEFRA, 2007; Himken et al., 1997). Fewer studies examine yield response to phosphate and K_2O (DEFRA, 2010).

A literature review by Cadoux et al. (2012), found that 6 studies out of 11 showed a yield response to N fertiliser application rates. These assume yield responses of 1-6 tonnes dry matter ha^{-1} a resulting from applications of 40-120 kg ha^{-1}. One study showed a benefit of 9.8 tonnes biomass ha^{-1} from applying up to 200 kg N ha^{-1} $year^{-1}$ (Ercoli et al., 1999). The yield response to fertiliser will be examined as part of a sensitivity analysis.

7.1.2.3. Nutrient Use Efficiency and Cycling in Miscanthus

The aim of fertiliser application is to ensure that the crop has sufficient nutrients for effective growth. Some studies suggest that fertiliser application should be sufficient to replace nutrient off-take in harvested biomass (Styles & Jones 2007; Styles et al. 2008). Estimating the off-take could be complicated by Miscanthus' ability to translocate nutrients to the rhizome prior to senescence during the winter (Jørgensen, 2011). Such translocation is suggested to give the plant its low-nutrient demanding traits (Cadoux et al., 2012). Recycling of K_2O is reported to be less efficient than N and P_2O_5, and a greater proportion of K_2O is removed in the harvested biomass (Cadoux et al., 2012). This may explain the higher K_2O-based fertiliser demands.

The translocation of N has been demonstrated by application of ^{15}N-labelled fertilisers (Christian et al., 2006) and nutrient translocation is reported to peak between June and October (**Figure 7-2**), with a lower content by the following spring (Cadoux et al., 2012).

It is suggested by Davis et al. (2010) that, despite translocation occurring, the N balance is not sufficient to replace the nutrients lost through Miscanthus harvest and removal from site. They suggest that the only method in which could supply sufficient N would be through some association of N-fixing microorganisms. They validate their theory through C and N modelling with the DAYCENT model, and find that a missing 25 g N m^{-1} $year^{-1}$ is unaccounted for. They conclude this must be provided through N fixation. Analysis of rhizome tissue identified *nifH*-positive bacteria strains, indicating the presence of dinitrogenase reductase (Davis et al. 2010). This would mean that N applications are either minimal, or unnecessary.

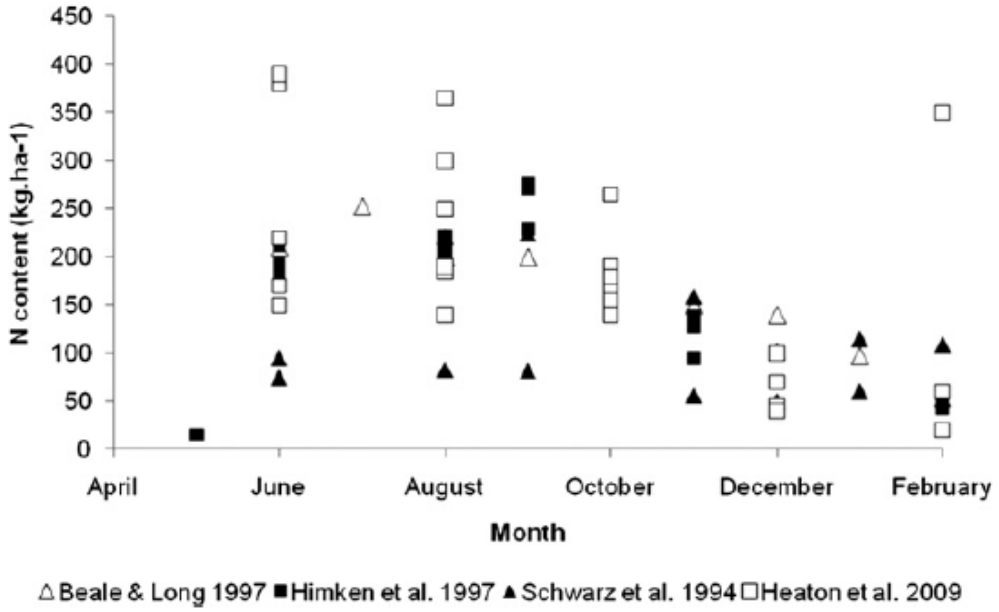

Figure 7-2. Changes in N content in the Miscanthus over the growing year in a review of literature performed by Cadoux et al. (2012).

7.1.2.4. Harvesting Time and Nutrient Demands

Due to the translocation of nutrients to the rhizome, it has been suggested that harvesting Miscanthus at an earlier time without compensating for nutrient off-take could exhaust the crop so that it is not viable for the full expected lifetime duration (Clifton-Brown et al., 2007). There is evidence that harvesting in the autumn can lead to an net higher off-take of 94 kg N, 15 kg P_2O_5 and 69 kg K_2O ha^{-1}, compared to harvesting in the spring (Clifton-Brown et al., 2007). Other field studies also anticipate that a larger fertiliser input would be required with an earlier harvest (Heaton et al., 2009).

The uncertainty analysis bases the contractors' accounts as the 'minimum' scenario and data from literature as the 'maximum' scenario. The following fertiliser regimes are suggested as suitable representations of conventional spring harvesting and early harvesting:

1) **Spring harvesting (conventional harvesting)**
    a. ***Establishment*** - the minimum scenario involves applying nothing to the establishing crop. A 'maximum' scenario of 130 kg N ha^{-1}, 130 kg P_2O_5 ha^{-1} and 300 kg K_2O ha^{-1} are assumed. This is according to literature review of the maximum application rate.
    b. ***Following years*** - a minimum single application of 40 kg K_2O after five years is suggested, according to information provided by contractors. A 'maximum' scenario of 100 kg N ha^{-1} $year^{-1}$, 50 kg P_2O_5 ha^{-1} $year^{-1}$ and 200 kg K_2O ha^{-1} $year^{-1}$ is assumed to be required to compensate for off-take of nutrients from the harvested crop. This is according to literature review of the maximum application rate.
2) **Autumn harvesting (advanced harvesting for bioethanol production)**
    a. ***Establishment*** – same as spring harvesting
    b. ***Following years*** – Due to lack of data, only a 'maximum' scenario, applications of 100 kg N ha^{-1} $year^{-1}$, 50 kg P_2O_5 ha^{-1} $year^{-1}$ and 200 kg K_2O ha^{-1} year are assumed.

Due to a lack of data, a uniform distribution must be assumed between minimum and maximum points. A sensitivity analysis will be performed in **Chapter 9** examining the impact of this assumption. Also, a sensitivity analysis will be performed to indicate the importance in determining fertiliser requirements of Miscanthus with a greater level of certainty.

On termination, it is suggested that nutrients must be replaced in the soil to compensate for nutrient uptake by the final removal of Miscanthus. A total soil debt of 102 kg P_2O_5 ha^{-1} and 527 kg K_2O ha^{-1} is suggested by Duffosé et al. (2012). This is assumed as a maximum because there is no indication on how much the plot studied by Duffosé et al. (2012) was fertilised during its lifetime (summarised in **Table 7-3**).

Table 7-3. Summary of fertiliser requirements for Miscanthus including autumn and winter harvesting.

Nutrient	Stage	Harvest Period	Application Rate (kg nutrient ha^{-1} $year^{-1}$)	
			Min	Max
N	Establishment	Autumn	0	130
		Spring		
	Following years	Autumn	100	
		Spring	0	100
P_2O_5	Establishment	Autumn	0	15
		Spring		
	Following years	Autumn	50	
		Spring	0	50
K_2O	Establishment	Autumn	0	300
		Spring		
	Following years	Autumn	200	
		Spring	40[a]	200
Termination				
P_2O_5			0	102
K_20			0	527

[a] *Applied once in year 5*

7.1.3. Fertiliser Scenarios

Two fertiliser scenarios for Miscanthus are selected based on the literature review. These will assume that the nutrients are supplied from:

- **Artificial fertilisers –** ammonium nitrate, triple superphosphate, potash
- **Organic fertilisers –** chicken litter (see discussion)

Many studies assume that only organic fertilisers will be used in energy crop cultivation (Felten et al. 2013; Gilbert et al. 2011; Smith & Slater 2010). These resources do not, however, state the quantity of fertiliser used. One study states only nutrients (Gilbert et al. 2011) while another only reports the GHG emissions from organic fertiliser application (Felten et al., 2013).

It is difficult to precisely measure the necessary application rate of organic fertilisers, as they contain more than one form of nutrient (Smith & Slater 2010). The limiting nutrient is always K_2O (Smith & Slater 2010). Applying a given rate of K_2O may mean that nitrogen application exceeds the 250 kg N ha^{-1} limit (Gilbert et al. 2011). There is also a limit of 250 tonnes ha^{-1} total organic matter that can be applied to land in the UK (Moffat, 2006). Gilbert et al. (2011) applied a lower concentration of K_2O to the Miscanthus crop than identified as necessary in this study (100 kg ha^{-1} every 3 years), yet even this application rate caused an excess application of 267 kg N ha^{-1}, which exceeds limitations in nitrate vulnerable zones.

In this study organic fertiliser rates are applied so N is not applied in excess. Chicken litter is identified as a suitable organic fertiliser for Miscanthus as it has a high dry matter content (71.5%) and a good ratio of nutrients to maximise those applied in organic form (Smith & Slater 2010). The application rates are given in **Table 7-4**. As K_2O is the lowest occurring nutrient in organic fertilisers, top up applications of K_2O are required to fulfil the nutrient demands of the crop. There are often excesses of N and P_2O_5 due to the concentration of nutrients in the organic fertilisers. Therefore, it is not possible to entirely regulate the nutrient application rate when applying organic fertilisers; yet such an issue is not discussed in those studies that apply it (Felten et al. 2013; Gilbert et al. 2011; Smith & Slater 2010).

Table 7-4. Application rates of organic fertilisers to Miscanthus.

Harvest Period	Stage	Application Rate			Excess Nutrients (kg nutrient ha^{-1})	
		Chicken Litter (tonnes ha^{-1})		'Top Up' of Artificial Fertilisers (kg nutrient ha^{-1})		
		Min	Max	K_2O	N	P_2O_5
Both	Establishment	0	7.6	0 – 130	(max: 120)	(max: 180)
	Termination	7.6		357	250	93
Autumn	Following years	7.6		30	250	145
Spring	Following years	1.8*	7.6	(max: 30)	(min: 59) (max: 250)	(max: 145)

** Applied once during 5th year*

The rates of application shown in **Table 7-4** suggest that it is unlikely that organic fertilisers would be applied in order satisfy the maximum fertiliser application rates listed in **Table 7-3**. Such an application rate would require vast quantities of organic fertilisers, and in some instances still require a "top up" of artificial fertilisers. This is examined as part of a sensitivity analysis.

7.1.4. N_2O Emissions from Soils

Emissions of GHG's from soil depend on the rate of N-based fertiliser application assumed. Direct N_2O emission rates for Miscanthus are expected to be the same as for arable crops, as demonstrated by experimental data (Drewer et al., 2012), although a conflicting study by Jorgensen et al. (1997) showed that N_2O emissions may be 2 to 3 times higher than in arable crops. Their study may have applied excessive N fertiliser, and may have applied it at the wrong time of year for the crop to absorb it (Cadoux et al., 2012).

Indirect N_2O emissions from Miscanthus are also expected to be the same for arable crops. One study showed an excessive leaching of 150 kg N ha^{-1} from unfertilised plots but suggested this may have arisen from the previous fertilised grassland that occupied the land (Cadoux et al.,

2012; Goulding et al., 2000). They demonstrate this by comparing unfertilised plots with fertilised plots and showed that excessive leaching only occurred in the latter in the second year of cultivation. After the first year, the authors recorded an average leaching of 30 kg N ha^{-1} $year^{-1}$ which is expected when applying between 150-200 kg N ha^{-1} (Goulding et al., 2000).

Crop N_2O emissions are based on a litter yield of 0.36 to 2.31 tonnes ha^{-1} (Riche, 2005), though only the smaller value is assumed for crops that are harvested in the autumn. It is suggested that N_2O emissions from Miscanthus residue addition to soil will be lower than that in an arable crop where the soil is turned over more frequently, although the decomposability of Miscanthus is expected to be the same as cereal straw (Foereid et al., 2004). The N_2O emission factor ranges specified in the IPCC are applied here in the calculations. Parameters for crop residues are provided in **Appendix 4**.

7.1.5. Pesticide Requirements

Herbicide applications are suggested at various stages of the growing cycle and the lifetime of the crop. Herbicide application after establishment is essential. Afterwards the mature crop is competitive and relatively fast growing, so applications are expected to be needed less frequently as the crop ages (Jørgensen, 2011).

Application rates range between 2.2 to 12.2 kg active ingredient (a.i) ha^{-1} for establishment (Smeets et al. 2009; Styles & Jones 2007). The first cut may also include an herbicide application of 3 to 4 kg a.i ha^{-1} to combat perennial weed growth (Bullard and Metcalf, 2001; Christian et al., 2008). Bullard and Metcalf (2001) suggest annual applications of 4 kg a.i ha^{-1}. Others suggest small applications that are made to target specific problem areas (Christian et al., 2008). The contractors reported application rates of 6 kg a.i ha^{-1} during establishment and 8 kg a.i ha^{-1} after the first cut, and then the leaf litter can effectively eliminate the need for weed control (I. Webber pers. com. 2012). Other pest and disease pesticides are not currently required (Bullard and Metcalf, 2001).

In this study, minimum application rates of herbicides are 4 kg a.i ha^{-1} during establishment, then a single application of 4 kg a.i ha^{-1} after first year maintenance, according to commercial-scale crops. Maximum application rates are assumed to be 10 kg a.i ha^{-1} on establishment, then 4 kg a.i ha^{-1} every year until termination; although it is acknowledged that this scenario is not carried out in commercial crops. An application of 1-2 kg a.i ha^{-1} glyphosate is required to terminate the crop (I. Webber pers. com. 2012, Gilbert et al. 2011).

7.1.6. Fuel Requirements

Data on fuel consumption rates is limited in most LCA studies. In this study, fuel consumption data and ranges were provided from two contractors. The fuel consumption is based on pure working rates and does not include idle time or waiting. Estimates for fuel consumption are estimated based on time records and their expert knowledge, therefore the accuracy of the data is as realistic as could be achieved without very detailed experimentation. The data is compared with that from literature as part of a validation process.

7.1.6.1. Site establishment

The following stages are included in site establishment in both rhizome crops and main Miscanthus crops.

- **Ploughing**
- **Power Harrowing**
- **Planting**
- **Rolling**
- **Herbicide application**

Site preparation involves both ploughing and power harrowing the site before the rhizomes are planted in rows. Contractor 1 estimated that ploughing requires an average of 22 litres ha^{-1}; ranging between 18 and 30 litres ha^{-1} depending on the soil type (I. Webber pers. com., 2012). Contractor 2 estimated a consumption rate of 15 litres ha^{-1} for ploughing (T. Barton pers. com. 2012). Power harrowing requires on average, 70 litres ha^{-1}, with little variation between soil types after they have been ploughed. Planting is carried out by a modified potato harvester. Planting requires, on average 30 litres ha^{-1}, again, with little variation with different soil types after a similar level of site preparation. Afterwards, the site is rolled to increase rhizome contact with the soil to prevent them from drying up. The site is sprayed with a pre-immergence herbicide. Rolling and spraying require 17 and 7 litres ha^{-1}, with little variation. The data is summarised in **Appendix 5**.

In total, 138-154 litres ha^{-1} of fuel are estimated by the contractor for site establishment. Other estimates in the literature are either lower: 61.2 litres ha^{-1} (Gilbert et al. 2011), 77 litres ha^{-1} (Smeets et al., 2009), 122 litres ha^{-1} (Monti et al., 2009), 93 litres ha^{-1} (Ercoli et al., 1999) or higher than the estimate: 198 litres ha^{-1} (Bullard and Metcalf, 2001). Differences in estimates could be due to different tractor fuel efficiencies, sizes and width of implements. The estimates in the literature contain the same activities as included in the commercial scale plot.

7.1.6.2. First Year Maintenance (Topping)

The first year maintenance phase involves cut back of the first year's growth to remove standing dead biomass and to encourage the further growth of the crop. In year 1, the cut biomass is left onsite. In subsequent years the biomass is removed and sold if economically viable. The process is performed in both rhizome crops and main Miscanthus crops. The mower consumes approximately 7 litres ha^{-1}. A fuel consumption rate of 8.5 litres ha^{-1} is estimated in the literature (Smeets et al., 2009).

7.1.6.3. Biomass Harvesting

Forage harvesting requires between 15 and 26 litres ha^{-1}, depending on the type of cut and the thickness of crop. In normal cutting the fuel requirement is 15 litres ha^{-1}. An estimate of 3.46 litres $tonne^{-1}$ is provided from the contractor for baling. Estimates from literature for harvesting range between 22.3 litres ha^{-1} (Smeets et al., 2009), 40.3 litres ha^{-1} (Styles & Jones 2007), 51 litres ha^{-1} (Bullard and Metcalf, 2001), 53 litres ha^{-1} (Monti et al., 2009).

7.1.6.4. Rhizome Harvesting

There is no fuel data available in the literature for rhizome harvesting, so the data is entirely provided from the contractor. The rhizome crop can be left for between 2 and 6 years, depending on how successful the stand grows, or rhizome demand (M. Carver pers. com. 2012). Rhizome harvesting occurs after biomass harvesting. The process involves some highly intensive operations that involve a high degree of interaction with soil. It is split into 6 stages:

- **Flail topping**
- **Flat lifting**
- **Rhizome lifting**
- **Rhizome sorting**
- **Rhizome storage**

Some of these stages are shown in **Figure 7-3**. First the above ground biomass is removed in two phases: a forage harvester is used to cut the main crop, leaving stubble (points 1 to 2). It is removed using a flail topper (points 2 to 3). The flat lifting phase (not shown) is used to loosen and lift the soil (point 4) so that a lifting machine (point 5) can pass over and harvest the rhizomes. Flat lifting requires the use of two machines. The first is a large subsoiler-like flat lift machine. The machine loosens the soil so that the second machine, a 'roto-spike' can be used to pick up rhizomes and leave them in a ridge. The final process of rhizome lifting is carried out by a modified potato harvester. The collected data is summarised in **Appendix 5**.

The non-uniform structure of rhizomes means that a large quantity of soil and stones are also lifted (point 6) and the material must be sorted at a designated sorting station, which is a short journey away from the field (2 miles). The transport distance could potentially be longer if the rhizome fields were further away, as the sorting station is not mobile, and there are few in the UK. Presently the scale of operation means that transport distances are small.

Figure 7-3. Example of Miscanthus rhizome harvesting phases.

Sorting involves tipping the rhizomes, soil and stones into a hopper where it is fed into a conveyor system. On the conveyor, rhizomes are sorted by hand. The electricity demands are reported at 5.5 kWh tonne rhizomes^{-1}. The material from the site contains about one third rhizomes, the rest being soil and stones. DEFRA regulations specify that the soil and stones must be returned to the site (point 7,RPA, DEFRA 2010).

The rhizome harvesting process takes place in spring. After sorting, rhizomes are loaded into a wooden crate or storage bags, and are transported for immediate planting. If there are delays in planting the rhizomes are stored in a warehouse. The warehouse is ventilated in order to stop the development of rhizomes. Storage is estimated to consume between 2.94 kWh tonne rhizomes^{-1} (M. Carver pers. com. 2012).

7.1.6.5. Crop Termination

Data was not available for crop termination as it is currently rarely carried out. In theory, the rhizome lifting process would not be performed on an old crop, and the rhizome lifting process in fact does not remove all rhizomes from the site. The contractors suggested that for complete eradication, a subsoil operation and high herbicide application would be sufficient for termination. Literature suggests a combination of biomass removal, or crushing, followed by herbicide applications and ploughing (Duffosé et al., 2012).

7.1.7. Use of Cultivation Machinery

The cultivation machinery is unlike the case study for wheat, as the machinery must be delivered to the site, as it is owned by a contractor. In total 10 implements and 6 machines are used during the Miscanthus growing cycle. Where the work rates are not given by the contractor, they are estimated based on similar processes in agriculture. A delivery distance of 50km is assumed, however a sensitivity analysis is performed on this assumption.

The GHG emissions from manufacture are based on the weights of the machines and implements, which are either taken from Williams et al. (2006), or estimated. The results are provided in **Appendix 3**. The total manufacture of the machinery is allocated to the time required to perform the given task set out to provide the functional unit of the study, assuming a total potential lifetime of 7000 hours (EcoInvent, 2007).

7.2. Summary of Inputs to Miscanthus

Miscanthus cultivation requires the input of diesel and pesticides. Diesel is required for site operations during both rhizome and Miscanthus cultivation. Pesticides are required during establishment, sometimes during maintenance, and during the termination event. There is high uncertainty in the fertiliser requirements of Miscanthus. There is evidence that autumn harvesting leads to a higher net off-take of nutrients therefore the fertiliser demands of Miscanthus increase when harvested in this period. It must be noted here that there is also some uncertainty in whether increasing nutrient application can effectively compensate for increased nutrient off-take in an autumn-harvested crop.

When applying organic fertilisers to Miscanthus, it is difficult to specifically apply a given amount of nutrient, specifically when considering is K_2O: the most important nutrient in Miscanthus. If organic fertilisers are used to reach a specific K_2O application rate there is a chance that excess application of nitrogen would be applied, potentially having negative implications on the total GHG emissions from organic fertiliser application. Variation in the inputs of Miscanthus are due to:

- Variability: Soil conditions affect diesel fuel consumption
- Uncertainty
    - Fertiliser requirements
    - Fertiliser types
    - Pesticide usage
    - Rotation length

7.3. Direct Land Use Change Effects

This section discusses the GHG emissions that may occur due to direct land use change (DLUC) in Miscanthus. DLUC is assumed to occur when Miscanthus is planted on arable land, grassland or forestland. Such a scenario is particularly relevant to the UK as it is a relatively new crop. As Miscanthus is usually grown on farmland, this is the focus of the further sub-sections. The conversion of grassland and forestland is calculated using RED/IPCC calculations and are listed in **Table 7-5**.

There are various issues involved in DLUC, therefore the following are examined:

- **Carbon sequestration rates according to literature** – discusses estimates from literature and considers how harvest date may impact sequestration
- **Direct land use change**– for conversion of grassland and forestland to Miscanthus.

7.3.1.1. Carbon Sequestration Rates According to Literature

This sub-section discusses SOC sequestration rates for when Miscanthus is grown on arable land. Although establishment may cause oxidation of soil organic matter through ploughing (Murphy et al. 2013), it is suggested in the literature that Miscanthus planted on arable land can increase the net carbon store of the soil (Hamelin et al., 2012; Hillier et al., 2009; St Clair et al., 2008), assuming the crop is left for a period of 20 years.

Hamelin et al. (2012) found that the extent of SOC change depends on the harvest season, soil type and climate. There is uncertainty in the total increase or decrease in SOC as it is determined by the amount of crop residues left in the field and their turnover-time (Foereid et al., 2004). During the lifetime of the crop, carbon additions to soil occur from leaves, rhizomes and shoots (Amougou et al., 2011), however the turnover rate of these residues is not clear (Hamelin et al., 2012).

Few studies examine the effect of harvest date on SOC sequestration rates. It is suggested that biomass losses during the winter period, are in fact SOC contributions (Amougou et al., 2011; Clifton-Brown et al., 2007). As these residues have high lignin contents, it suggests that they would enter the less degradable and hence-longer lasting carbon pool (Amougou et al., 2011). In this case, earlier harvesting may reduce the return of residues to soil and limit the rate of carbon sequestration achieved. Hamelin et al. (2012) discovered a linear relationship between biomass decomposition and SOC sequestration and found a decreased decay rate of 25% lead to an increase of 45% in SOC where biomass was more permanently incorporated into soil. It is possible that a similar decrease in biomass returned to the soil will lead to a similar decrease in SOC sequestered. A similar result by Grogan & Matthews (2001) showed that SOC sequestration was reduced by 47% between winter and spring harvesting (Mishra et al. 2012).

For conventional harvesting in spring, estimates for SOC sequestration in the literature range between a minimum and maximum sequestration rate of 0.2 – 5.5 kg CO_2 eq. ha^{-1} $year^{-1}$. It is assumed that the rate of sequestration is constant during rotation. This assumption is supported by the results of studies, where there is a very weak (R^2 = 0.1) correlation between year in which the SOC measurement was made and the rate of SOC sequestration (**Figure 7-4**).

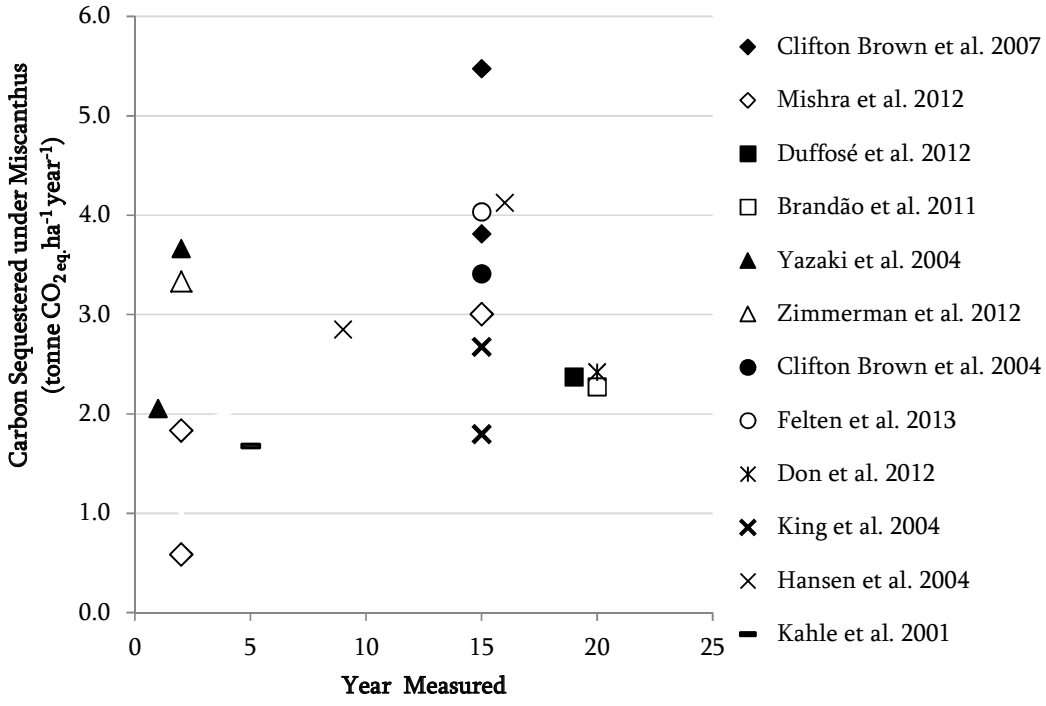

Figure 7-4. Review of carbon sequestration rates under Miscanthus from literature.

Rarely mentioned in studies of carbon storage under Miscanthus, is the extent of which the carbon is retained when the plant is terminated (Clifton-Brown et al., 2007), mainly because few studies cover sufficient time periods. A study by Duffosé et al. (2012) terminated a 20-year old stand of Miscanthus in France by crushing above-ground biomass, applying herbicide and ploughing. They estimated a sequestration rate of 45.1 tonnes CO_2 eq. ha^{-1}, representing an increase of 9.6% compared to annual land. After termination they estimated a total emission of 15.4 tonnes CO_2 eq. ha^{-1} from bare soil, indicating a loss of 34% of the stored carbon due to oxidation of SOC during the termination phase.

7.3.1.2. Conversion of Grassland and Forestland

Negative changes may occur if forestland is converted to Miscanthus (Don et al., 2012), as these systems contain, at equilibrium, a higher net carbon content in the soil (Hillier et al., 2009). The IPCC and RED calculations are used to calculate these SOC losses. The results are summarised in **Table 7-5**.

Table 7-5. Summary of GHG emissions from DLUC to Miscanthus.

Land Type	Harvest Date	Change with Conversion to Miscanthus (tonnes CO_2 eq. ha^{-1} $year^{-1}$)	
		Min	Max
Forestland	N/A	-1.9	
Grassland	N/A	0.1	
Arable Land (IPCC)	N/A	4.4	6.9
Arable Land (sequestered)	Autumn	0.1	2.9
	Spring	0.2	5.5
Arable Land (lost with termination)	Autumn	0.03	1.0
	Spring	0.1	1.9
Arable Land (net gain)	Autumn	0.1	1.9
	Spring	0.1	3.6

7.4. Outputs from Miscanthus Cultivation

The following section describes the material outputs from Miscanthus cultivation. The following outputs are discussed here:

- **Rhizome yields from propagation**
- **Main crop Miscanthus yields**

These are described in the following sub-sections.

7.4.1. Rhizome Yield

Rhizome yields are not well reported in the literature, as data on the rhizome propagation phase in general are limited. Instead, a multiplication ratio of 1:3 (Atkinson, 2009) to 1:20 (Bullard and Metcalf, 2001) is reported. Based on the data from contractors, for every 2 tonnes input of rhizomes, an output yield of 28 tonnes of rhizomes is achieved, or a multiplication ratio of 1:14. A lower estimate of 20 tonnes ha^{-1} is estimated by Bullard & Metcalf (2001). Therefore 0.01 to 0.05 hectares of a rhizome crop are needed for one hectare of Miscanthus main crop.

7.4.2. Miscanthus Yield

There are various assumptions that must be made when determining the yield of Miscanthus. The first major issue is determining the lifetime of the crop. The GHG emissions from establishment and termination are shared between every tonne of Miscanthus that leaves the site over its lifetime.

The following discussion is structured into the following sub-sections:

- **Geographic variation in Miscanthus yields**
- **Temporal variation in Miscanthus yields**
- **Effect of harvesting time on Miscanthus yields**

7.4.2.1. Geographic variation in Miscanthus yields

Variation in Miscanthus yields has been attributed to climatic conditions, soil water availability, nutrients availability, plant density, harvest time and (Danalatos et al., 2007). Miscanthus yields vary greatly in the literature, ranging from 2 to 44 tonnes ha^{-1} (Lewandowski et al., 2000). The European Miscanthus Productivity Network reports yields ranging from 7.7 tonnes ha^{-1} to 26.3 tonnes ha^{-1} in 3-year-old crops (Christian et al., 2008).

Field-based trials lasting 15 years in the UK have shown yields of 8.1 and 16.2, or an average of 12.8 with a standard deviation 2.9 (Richter et al., 2008). Other modelled estimates based on water-based limitations estimate yields of 6.9 to 24.1 tonnes ha^{-1} $year^{-1}$ (Price et al., 2004). The contractor reported an average yield of 8.0 tonnes ha^{-1} on four 6-year old commercial stands in South West England (T. Barton pers. com. 2012). The UK average is predicted to be 10.45 ODT ha^{-1} $year^{-1}$ (Wang et al. 2011), or 13.9 tonnes ha^{-1} $year^{-1}$ at a 25% moisture content.

7.4.2.2. Temporal variation in Miscanthus yields

Due to the nature of Miscanthus' growth, there is also a temporal aspect to the yield (**Figure 7-11**). The 'top' yield is typically not achieved until year 3-5 once the crop has become fully established (Lewandowski et al., 2000). Termination is assumed to occur when the crop yields decrease due to the crop reaching the end of its life (Jørgensen, 2011), however the exhaustion of the crop is yet to be demonstrated in field trials. A trial in Ireland showed that after 15 years, yield losses began to occur, that was not predicted by yield models (Clifton-Brown et al., 2007).

Another study showed that yields peaked at 17.7 tonnes ha^{-1} in the 10th year (Christian et al., 2008). There is evidence that stands as old as 20 years can have an average yield of 14 tonnes ha^{-1} (Duffosé et al., 2012).

This study assumes a minimum and maximum lifetime of five (Natural England, 2013a) to 20 (Bullard and Metcalf, 2001) years for a Miscanthus crop. If the crop was supported under the Energy Crop Scheme then the farmer must commit to maintaining the crop for a minimal of five years, otherwise they must repay the establishment grant (Natural England, 2013a).

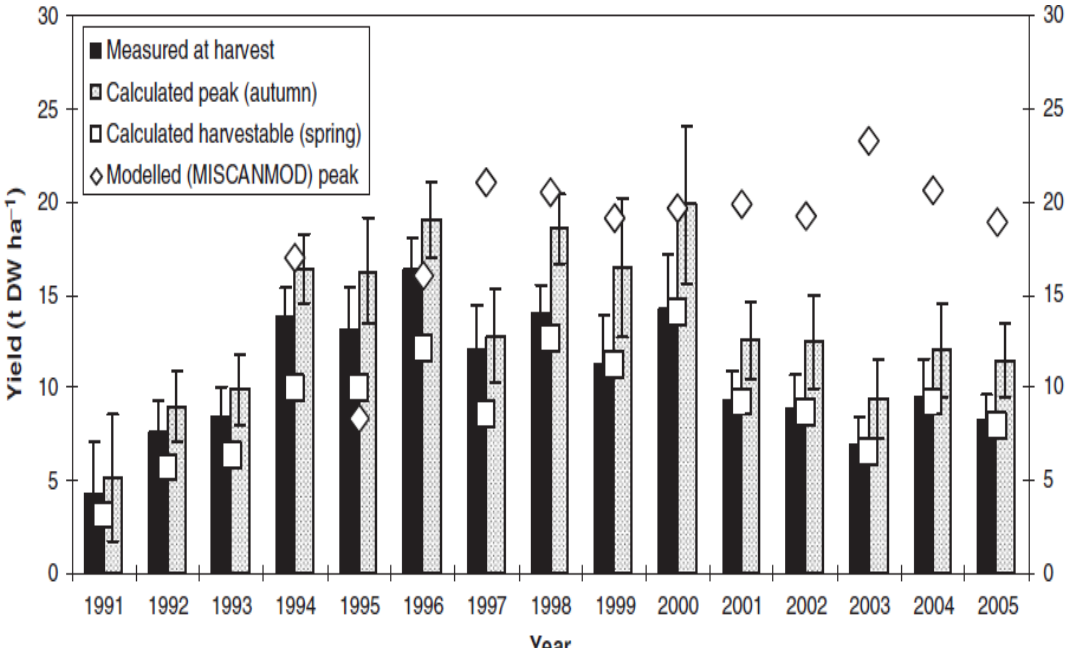

Figure 7-5. Yield patterns of Miscanthus over 15 years (Clifton-Brown et al., 2007).

7.4.2.3. Effect of harvesting time on Miscanthus Yields

There are also other, smaller temporal aspects to the yield of Miscanthus. It is shown in the literature that yield losses of up to 4.4 tonnes ha^{-1} occur between the winter and following spring, representing a loss of one third of the top yield (Clifton-Brown et al., 2007). This is due to a combination of the drying of the crop and dry matter losses over the winter period (Lewandowski and Heinz, 2003). Losses of 3 tonnes ha^{-1} were also observed by Amougou et al. (2011) and the authors suggest they are necessary for nutrient and carbon recycling.

There is a trade-off between the yield and moisture content of the biomass (Danalatos et al., 2007). Drier biomass is more suitable for heat and power production, but may require primary energy consumption for drying unless the crop is harvested in the spring (Lewandowski and Heinz, 2003).

This issue is complicated by the translocation of nutrients to the rhizome and overwintering phase. Translocation is said to improve the combustion characteristics of the biomass as in spring Miscanthus contains less chloride, sulphur, nitrogen, potassium and ash, which may cause corrosion and slagging in biomass boilers (Lewandowski & Kicherer 1997). Similar results are also found in the perennial grass *Phalaris arundinacea*, or reed canary grass (Landström et al., 1996). Additionally, the translocation of nutrients is believed to be the reason that Miscanthus is a low-input crop (Cadoux et al., 2012). Harvesting before this point may increase the nutrient demands of the crop, otherwise it may become exhausted prematurely (Clifton-Brown et al., 2007).

7.4.2.4. Summary

From the above sub-sections, the two harvesting scenarios are presented:

1) **Spring harvesting (conventional harvesting):** top yields ranging 8.1 and 16.2 tonnes ha^{-1} $year^{-1}$, or an average of 12.8 tonnes ha^{-1} $year^{-1}$.
2) **Autumn harvesting (advanced harvesting for bioethanol production):** assumed to be 150% of the spring yield, which represents a loss of one third over the winter period (Clifton-Brown et al., 2007; Lewandowski and Heinz, 2003).

It is assumed that the first years' harvest is zero, and the second year harvest is 50% of that of the top yield as demonstrated by Bullard and Metcalf (2001) and Himken et al. (1997). These yields are summarised in **Table 7-6**.

Yield is assumed to be normally distributed (Richter et al., 2008). It is assumed that the yield of spring harvests is approximately one third lower than those in the autumn. The average yields over a five and 20 year rotation period are calculated according to a Monte Carlo Analysis using 1000 runs.

Table 7-6. Average yields for Miscanthus with autumn and spring harvests.

Details		Units	Autumn Harvest		Spring Harvest	
		tonnes	Average	S.D	Average	S.D
Yield Details	Year 1		0	0	0	0
	Year 2	ha^{-1}	9.6	2.1	6.4	1.4
	Year 3 +	ha^{-1}	18.9	4.5	12.8	2.9
Average Yield Over Rotation	5 Years	ha^{-1} $year^{-1}$	13.4	3.0	9.0	2.0
	20 Years	ha^{-1} $year^{-1}$	17.5	4.1	11.8	2.7
Total Yield Over Rotation	5 Years	ha^{-1} $rotation^{-1}$	66.4	15.4	44.5	10.3
	20 Years	ha^{-1} $rotation^{-1}$	350.7	81.4	234.4	54.6

7.4.3. Yield and Rotation Length

The total output from a site of Miscanthus will depend on the time period in which the crop is in 'production'. The rotation length is assumed to be between five and 20 years, so there is potentially a large range in total net outputs between these two time periods. Although there are no co-products in Miscanthus cultivation, there is some temporal allocation in the establishment and the termination phases. For example, the GHG emissions from site establishment are split between every tonne of Miscanthus that is harvested over the productive time of the crop (**Figure 7-6**). Therefore, as the rotation length increases the GHG emissions from establishment and termination become smaller on a 'per tonne' basis, or a 'per MJ' basis of bioethanol. Harvesting impacts are not shared as they are on a per-tonne basis. The results will be used to test the sensitivity of the GHG emission results to rotation length.

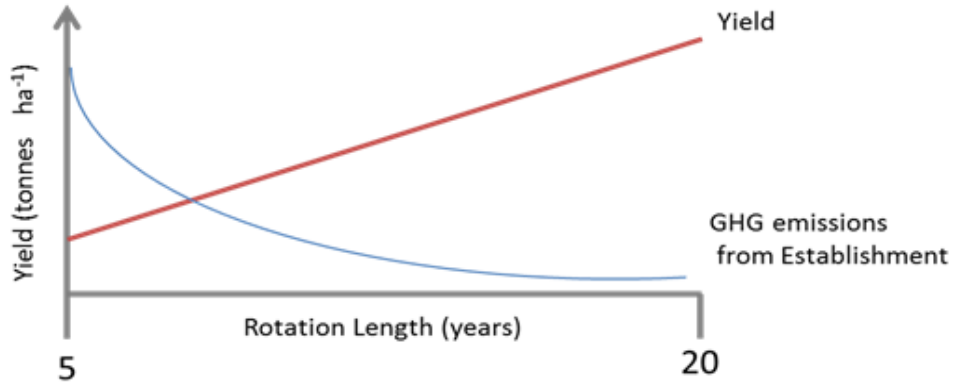

Figure 7-6. Predicted effect of GHG emissions from establishment with increasing rotation period of Miscanthus.

7.5. Summary of Miscanthus Cultivation

Miscanthus is a perennial energy crop that is harvested annually for biomass production. In the literature relatively high yields, low nutrient demands and ability to sequester carbon are expected to attribute the crop with a low overall GHG balance (Hillier et al., 2009). In this study, it has become apparent that there is a great deal of uncertainty with regards to the overall fertiliser requirements and yields from Miscanthus. Overall, there is uncertainty and variations in:

- Effects of harvest time on the GHG emission results
- Fertiliser requirements and frequencies of applications
- Fertiliser types applied
- Carbon sequestration, plus that remaining after termination
- Rotation length of the crop.

The following section examines the bioethanol production processes.

7.6. The Bioethanol Model

This chapter describes the model dataset collected for the bioethanol case studies. The case study aims to represent current knowledge on the conversion processes of 1st and 2nd generation bioethanol production. The conversion of wheat grain to bioethanol has been studied in various publications; however the same cannot be said for wheat straw and Miscanthus. **This study therefore provides a novel account of the GHG emissions of bioethanol production from Miscanthus. The application of the results to the current LCA accounting rules in the GHG reporting methodologies is also novel.**

An overview of these processes is provided in **Chapter 4** and summarised in **Figure 7-7**. The aim is to use the case study to estimate the GHG emission savings of using wheat grain, wheat straw and Miscanthus as a bioethanol source. The variability of these GHG emission savings will be assessed according to the variability of the dataset as well as the LCA methodology applied.

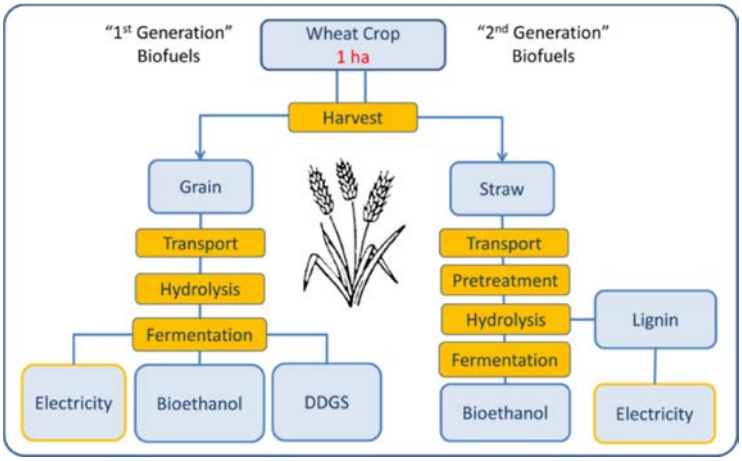

Figure 7-7. An overview of 1st and 2nd generation biofuel production with wheat.

This Chapter shows the results of the inventory data collection for the bioethanol production process, including a discussion of the inputs to the process, including heat and power and chemicals, and outputs of bioethanol and co-products (**Table 7-7**).

Table 7-7. Inputs, outputs from 1st and 2nd generation bioethanol production.

1st Generation Bioethanol		2nd Generation Bioethanol	
Inputs	**Outputs**	**Inputs**	**Outputs**
• Biomass	• Bioethanol	• Biomass	• Bioethanol
• Chemical reagents	• DDGS	• Chemical reagents	• Lignin
• Biological reagents	• Electricity	• Biological reagents	
• Fuel		• Fuel	

7.7. First Generation Bioethanol Production

The 1st Generation bioethanol production process is described in **Chapter 4**. In summary, the process utilises sugars and starch as a feedstock. Conventional fermentation technology is used to ferment these to ethanol. The conversion process includes transportation of grain, milling, hydrolysis and fermentation and distillation to bioethanol. **Figure 7-8** shows an overview flow diagram of the 1st generation bioethanol production process.

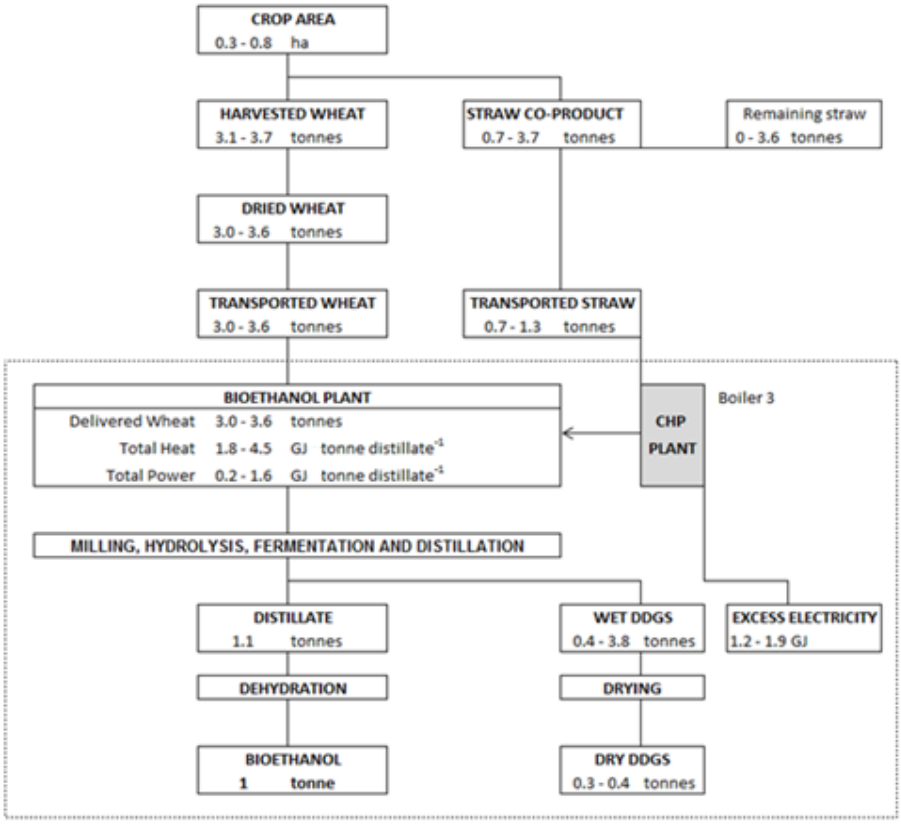

Figure 7-8. Overview of inputs and outputs to 1st generation bioethanol production utilising a straw-based CHP boiler.

7.7.1. Inputs to 1st Generation Bioethanol Production

Here the inputs associated with 1st generation bioethanol production are identified. Inputs are defined as materials that are delivered to the bioethanol plant and consumed in the conversion process. The main inputs are:

- **Biomass**
- **Chemical and biological reagents**
- **Fuel**

Data was collected from a range of studies (such as Bernesson et al. 2006; Clarke et al. 2008; Malça & Freire 2006; Mortimer et al. 2004; Punter et al. 2004; Weinberg & Kaltschmitt 2013) and four calculation tools: BEAT (AEA Technology and North Energy Associates, 2010), the HGCA Bioethanol Calculator (HGCA, 2011), Biograce (Biograce.net, 2012), and the Renewable Fuel Association's Solid and Gaseous Biomass Calculator (Westphal et al., 2011). There are more LCA studies in the literature though they lack transparency (Gnansounou et al., 2008; Yan and Boies, 2013). Transparent studies provide heat, power and chemical requirements for major processes, as well as yields of co-products. Many of the literature resources use only hypothetical data. Gnansounou et al. (2008), Martinez-Hernandez et al. (2013) and Punter et al. (2004) utilise data from industrial sources, though only the last provides transparent data.

7.7.2. Biomass

There are two types of biomass inputs to bioethanol plants:

- **Wheat grain** – provides the starchy substrate for bioethanol production
- **Wheat straw** – can be used for heat and power generation

The yield of bioethanol from one tonne of wheat must be determined in order to determine the biomass requirement to satisfy the functional unit of "1 tonne of bioethanol, to the 'factory gate', where it is ready to be blended and used as a fuel".

Bioethanol yields average 310 kg tonne grain^{-1}, ranging between 280 to 340 kg tonne grain^{-1}. Dry grain contains approximately 72% starch and sugar, and the maximum stoichiometric yield of bioethanol is 352 kg tonne straw^{-1} at 14% moisture content (Clarke et al., 2008). The yields reported in the literature review represent a process with 78-91% conversion efficiency. Therefore, the functional unit requires between 2.9 and 3.6 tonnes wheat, and requires between 0.2 and 0.8 hectares.

Such an area of land will provide between 0.7 and 5.5 tonnes of straw. When used as a source of fuel to provide heat and power for the conversion process it is assumed that both the wheat grain and straw are transported from the field to the plant. The energy content of straw is estimated at 13-14 GJ tonne^{-1} (**Table 6-13** of **Chapter 6**), and the straw supply from tonne of bioethanol may range between 9.7 and 52.8 GJ.

Table 7-8. Summary of biomass requirements for 1st generation bioethanol production.

Feedstock	Bioethanol Yield (kg tonne^{-1})		Land Required (ha)	
	Min	Max	Min	Max
Wheat Grain	230	340	0.2	0.7

7.7.3. Chemical and Biological Reagent Demands

The chemicals required for bioethanol production from wheat grain are listed in **Chapter 4, Section 4.5**. Only two references provide sufficiently transparent data on chemical reagents that are required for wheat grain conversion to bioethanol (AEA Technology and North Energy Associates, 2008; Bernesson et al., 2006). These are listed below in **Table 7-9**.

Table 7-9. Summary of chemical requirements for 1st generation bioethanol production.

Description	Value (kg tonne wheat input^{-1})		
	Average	Minimum	Maximum
Caustic Soda (49%)	13.88[a]		
Diammonium Phosphate (21%)	9.16[a]		
Sulphuric Acid (93%)	5.63	2.12[b]	9.13[a]
Enzyme AMG	7.9 [a]		
Enzyme α amylase	4.5 [a]		
Calcium Chloride	0.63	0.3[a]	0.95[b]
Yeast	0.09[b]		

a. (AEA Technology and North Energy Associates, 2010)

b. (Bernesson et al., 2006)

7.7.4. Fuel Requirements

The conversion process requires both heat and power, particularly for processes that heat water such as cooking, distillation and DDGS drying (Clarke et al., 2008). There are modest electricity requirements for crushing and auxiliary management of equipment (**Figure 7-9**).

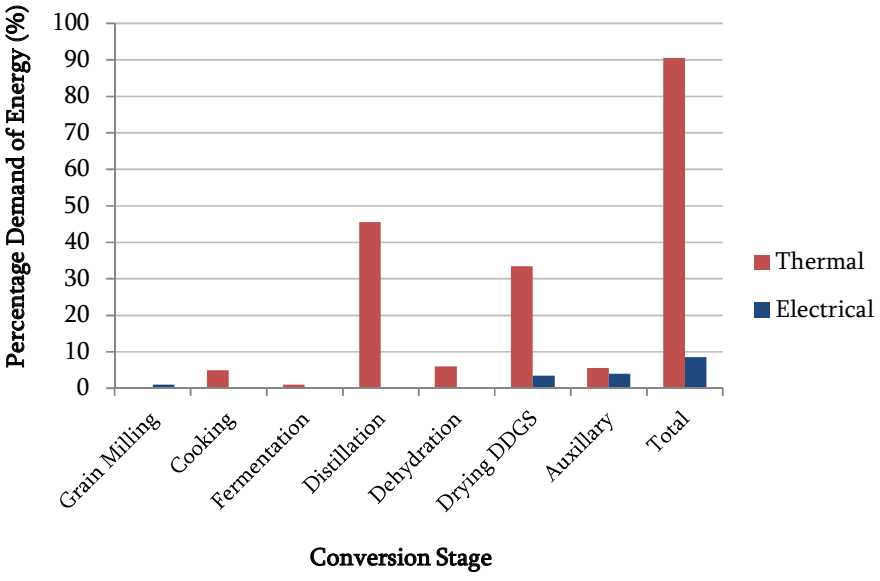

Figure 7-9. Energy requirements for bioethanol production (Clarke et al. 2008).

A combined heat and power (CHP) plant can reduce the overall energy requirements by 15% to the 'outdated practice' of utilising separate natural gas boilers and taking electricity from the grid (EUCAR et al., 2006; Punter et al., 2004). In the literature review, only two studies assumed a separate boiler and imported electricity (Bernesson et al., 2006; Malça and Freire, 2006).

The CHP plant can be powered with natural gas, coal, straw or DDGS (EUCAR et al., 2006; Punter et al., 2004). Using biomass can, in theory, give greater GHG savings. Therefore, in this study two CHP units are assumed to be utilised, being powered by natural gas and straw.

There is a great variation in estimates for the energy demand by bioethanol plants (**Table 7-10**), leading to different GHG emission results for conversion between studies (Yan and Boies, 2013). Heat and power demands may depend on the scale of the bioethanol plant (Bernesson et al., 2006) or the technology used (Punter et al., 2004). The majority heat requirements are for distillation and DDGS drying, whereas electricity is mainly required for fermentation and drying DDGS (Bernesson et al., 2006). One reference provided an energy requirement of 36 MJ tonne wheat^{-1} for waste water treatment (Bernesson et al., 2006).

Table 7-10. Primary energy requirements for 1st generation bioethanol (GJ tonne wheat^{-1}).

Stage	Average	Minimum	Maximum
Heat for conversion	3.11[a]	1.75b	4.52c
Electricity for conversion	0.58[a]	0.17b	1.61d
Electricity for waste water treatment	0.03[e]	0.03	

a. *Average from literature*
b. (AEA Technology and North Energy Associates, 2010)
c. (Biograce.net, 2012)
d. (Malça and Freire, 2006)
e. (Bernesson et al., 2006)

The CHP unit generates both heat and electricity. In some cases in the literature (AEA Technology and North Energy Associates, 2008; Mortimer et al., 2004; Punter et al., 2004; Yan and Boies, 2013), there is excess electricity to that required by the process, which is exported to the national grid. The ratio of heat and power produced can be configured within the boundaries of the technology to optimise the electricity generation rate of the process for economic reasons (Aylott et al., 2012). The quantity of electricity generated will depend on the technology utilised and the incentives to produce extra revenue from exported electricity (Aden et al., 2002).

Punter et al., (2004) provide some example model CHP plants that could be used. These are:

- **Boiler 1:** Natural gas-fired turbine, plus backpressure steam turbo-generator
- **Boiler 2:** Natural gas-fired turbine, co-fired HRSG, plus backpressure steam turbo-generator
- **Boiler 3:** Straw-fired boiler, plus backpressure steam turbo-generator

Further details, including efficiencies and heat to power ratios are provided in **Table 7-11**. It is assumed that the size of the CHP plant is matched to the heat demands for the process. It is assumes that the bioethanol plant has a total working life of 8064 hours year^{-1}, and a total output of 100,000 tonnes bioethanol year^{-1} (AEA Technology and North Energy Associates, 2010). Therefore, based on the bioethanol yield and a heat demand of 1.75 to 4.25 GJ tonne wheat^{-1}, the CHP plant will be sized between 13 and 44 MW (**Table 7-11**). The bioethanol plant in BEAT is sized at 34 MW.

Table 7-11. Model CHP plants used in this study (Punter et al., 2004).

Boiler	Overall Thermal Efficiency (%)	Heat to Power Ratio	Rating (MW)		Exported Electricity (kWh tonne bioethanol^{-1})	
			Min	Max	Min	Max
Boiler 1	70	1.77	13	35	0.79	0.91
Boiler 2	84	1.47	12	31	0.99	1.43
Boiler 3	64	1.26	17	44	1.20	1.94

7.7.5. Outputs from 1st Generation Bioethanol Production

There are three main outputs from 1st generation bioethanol production from wheat grain:

- **Bioethanol**
- **DDGS**
- **Electricity**

The functional unit of this study is to produce 1 tonne of bioethanol, to the 'factory gate'; and the outputs of DDGS and electricity are discussed in the following sub-sections.

8.1.1.1. DDGS

Wheat grain contains 60-70% starch. The remaining protein and fibre components cannot be converted to bioethanol (Weinberg and Kaltschmitt, 2013). These residues form a co-product known as dry distillers grain and solubles (DDGS). As of January 2008, DDGS can be dried and sold for use as animal feed or burnt for energy (Environment Agency, 2008). It is possible to obtain more valuable products such as bran and gluten by processing prior to the hydrolysis phase (Weinberg and Kaltschmitt, 2013), however this is not explored in detail here as no bioethanol plants are known to practice it (Clarke et al., 2008).

Yields of DDGS range between 0.3 and 0.4 tonnes tonne wheat^{-1}. Most studies report a yield of 0.4 tonne, although these estimates originate from the same study (Mortimer et al., 2004). The reported yield for DDGS is for that which has been dried to 10% moisture content so that it is ready to be sold for animal feed or energy (Punter et al., 2004). DDGS leaves the fermentation tank at between a 40% (Aylott et al., 2012) and 90% (Bernesson et al., 2006) moisture content.

8.1.1.2. Electricity

Studies utilising a CHP unit reported an average excess electricity generation of 3.74 GJ tonne wheat^{-1}, ranging between 0.41 to 2.54 GJ tonne wheat^{-1} (AEA Technology and North Energy Associates, 2008; Biograce.net, 2012; Mortimer et al., 2004; Punter et al., 2004; Westphal et al., 2007). The heat to power ratio ranged from 1.21 to 1.93, depending on technology (AEA Technology and North Energy Associates, 2008; Punter et al., 2004).

In this study, exported electricity is calculated based on the model CHP plants listed in **Table 7-11**, combined with the heat and power requirements per tonne bioethanol, listed in **Table 7-10**. A total exported electricity of 0.79 to 1.94 GJ tonne bioethanol^{-1} is estimated. An example of the requirements and outputs from the boilers (for CHP boiler Model 1) is shown in **Figure 7-10**.

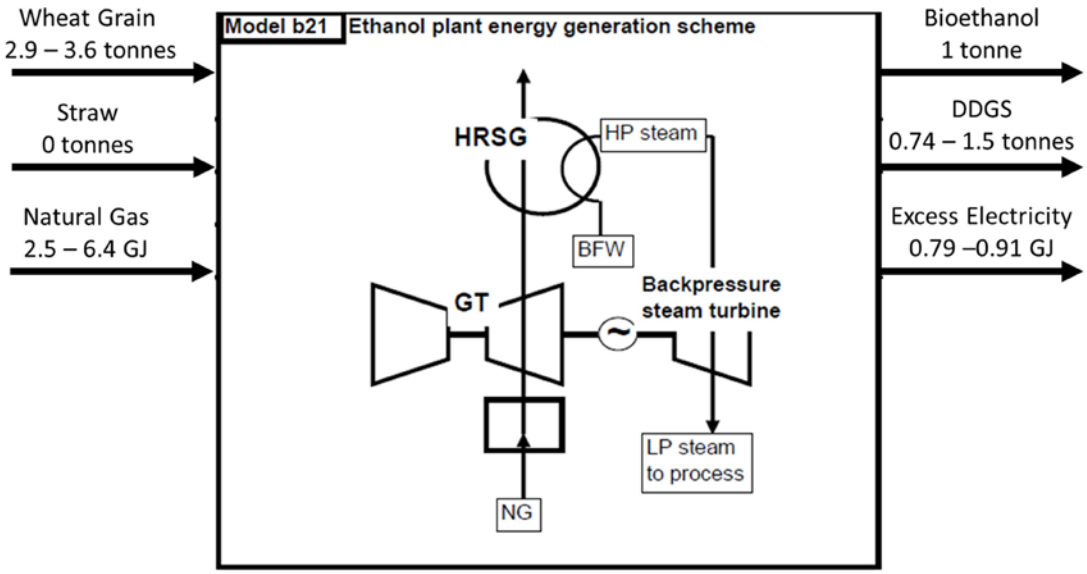

Figure 7-10. Example of input and output from CHP boiler Model 1.

7.7.6. Summary of Outputs of 1st Generation Bioethanol Production

The main outputs from 1st generation bioethanol are: bioethanol, DDGS and exported electricity. **Table 7-12** summarises the quantities involved per tonne of wheat input. Approximately 2.9 to 3.6 tonnes of wheat are required per tonne bioethanol, which requires between 0.2 and 0.7 hectares of arable land. On such an area, between 0.7 and 3.7 tonnes of straw is produced, which can feasibly provide between 9.7 and 52.8 GJ to the conversion process.

Based on the heat and power requirements listed in **Table 7-10**, straw provides sufficient inherent energy for the process, and there will be an excess of 0.1 to 3.7 tonnes of straw that could either be used for heat and power production for another process, or left on the site.

Table 7-12. Summary of the outputs from 1st generation bioethanol from wheat grain.

Description	Units (tonne wheat^{-1})	Value		
		Average	Minimum	Maximum
Bioethanol yield	tonne	0.31	0.28	0.37
DDGS (wet)				
DDGS (dry)		0.4	0.25	0.42
Exported electricity	GJ	0.38	0.22	0.66

7.8. Second Generation Bioethanol Production

This section describes the 2nd generation bioethanol production system, which involves either the microbial or biochemical conversion of wheat straw and Miscanthus to bioethanol (Singh et al. 2010). The process utilises a biochemical, or enzymatic conversion process. **Figure 7-11** provides an example of a 2nd generation bioethanol production process that utilises wheat straw.

Data was collected from five main studies (Aden et al., 2002; Borrion et al., 2012b; Slade et al., 2009; Wang et al., 2012; Wooley et al., 1999). Many of the studies lack transparency though can be used in some way to support the assumptions made by more detailed studies. For consistency between literature resources, this study examines a lignocellulosic conversion process utilising diluted acid pre-treatment followed by enzymatic hydrolysis. This is a suitable assumption for the first wave of lignocellulosic plants (Hamelinck et al. 2005a). All of the literature resources examined here use hypothetical data from process-based models.

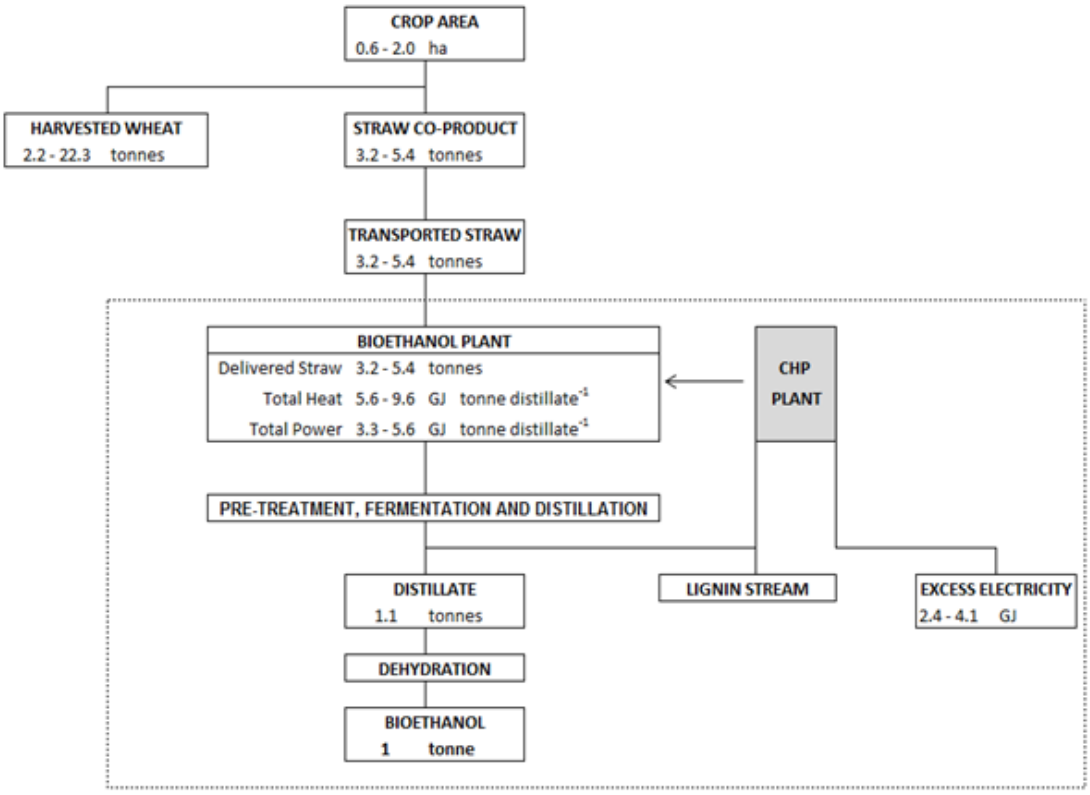

Figure 7-11. Overview of 2nd generation bioethanol process assuming excess electricity.

7.8.1. Inputs to 2nd Generation Bioethanol Production

Here the inputs associated with 2nd generation bioethanol production are identified. Inputs are defined as materials that are delivered to the bioethanol plant and consumed in the conversion process. The following are identified:

- Biomass – wheat straw or Miscanthus
- Fuel – provided by lignin and imported electricity
- Chemical and biological reagents

7.8.1.1. Types of Biomass

Similarly to the 1st generation bioethanol process, there are two types of biomass inputs to bioethanol plants, one that is the bioethanol feedstock, and another that can be used as a source of fuel:

- Lignocellulosic material - provides the sugars for bioethanol production
- Lignin – can be used for heat and power generation

Wheat straw and Miscanthus are two types of lignocellulosic biomass that are used in these case studies. The following section discusses the quantities required to fulfil the functional unit.

7.8.1.2. Quantities of Biomass

The functional unit of this study is to produce 1 tonne of bioethanol, to the 'factory gate', where it is ready to be blended and used as a fuel. Lignocellulosic bioethanol yields can be uncertain (Wang et al. 2012). Experiments with 'generic' lignocellulosic biomass report average yields of 379 kg tonne^{-1}, ranging between 358-400 kg bioethanol from one tonne of biomass (Spatari et al., 2010). Bioethanol yields are reported for specific technologies and biomass types. The overall conversion efficiency for enzymatic hydrolysis of combined cellulose, glucose, xylose and other sugars is reported to be between 75-85% of the theoretical maximum, with expected improvements to 85-95% in the future (Hamelinck et al. 2005a).

The theoretical yield depends on the specific sugar, cellulose, hemicellulose (etc.) composition of the biomass (Hodgson et al., 2010; Yoshida et al., 2008), as well as the pre-treatment method used (Sørensen et al., 2008). For wheat straw the theoretical bioethanol yield is 366 kg dry tonne^{-1} (**Table 7-13**). A more recent review reports a yield of between 185 and 316 kg tonne straw^{-1} (Borrion pers. com. 2012), representing 51% to 86% conversion efficiency.

The bioethanol yield of Miscanthus can be highly uncertain due to a combination of processing techniques as well as plant genotypes (Hodgson et al., 2010). Some estimates are vague, such as a yield of 4,600 to 12,4000 litres ha^{-1} from a yield of 14-40 tonnes ha^{-1} (Byrt et al., 2011). Even recent reviews claim to not have sufficient data to determine specific yields from grasses (Wang et al. 2012). In this study, the Miscanthus yield is estimated based on compositional data provided by Brosse et al. (2012) and using an online Theoretical Bioethanol Yield Calculator (EERE, n.d.). It is therefore possible to estimate theoretical bioethanol yields for autumn and spring harvesting (**Table 7-13**). A similar conversion efficiency as wheat straw is assumed (35%-75%) to deduce the bioethanol yield. Based on compositional data, a spring harvest is expected to give an 11% higher bioethanol yield than harvesting in the autumn.

Table 7-13. Bioethanol yields from Miscanthus and wheat straw.

Component (%)	Miscanthus[b]		Wheat Straw[d]
(Harvest Period)	Autumn	Spring	
Glucose	38	39.5	38.8
Galactose	0.3	0.4	2.7
Mannose	0[c]	0[c]	1.7
Xylose	14.9	19	22.2
Arabinose	1.1	1.8	4.7
Theoretical Yield[a]	**271**	**301**	**366**
51% of Yield (Min)	**143**	**160**	**185**
86% of Yield (Max)	**244**	**273**	**316**

a) *Units is kg bioethanol tonne dry matter^{-1}*
b) *Data provided from* Brosse et al. (2012)
c) *No data in* Brosse et al. (2012) *however this is expected to be zero or negligible* (Villaverde et al., 2010).
d) *Data provided from* Erdei et al. (2010)

7.8.2. Chemical and Biological Reagents

Chemical requirements for pre-treatment and conversion are listed in **Section 4.6** of **Chapter 4** and are summarised in **Table 7-14**, and are a combination of results of Borrion et al. (2012), Slade et al. (2009) and Wang et al. (2012).

Enzyme and yeast requirements are provided by Wang et al. (2012). It is assumed that enzyme manufacture takes place externally. GHG Emissions from cellulose enzyme manufacture must be calculated as these are not directly stated in Borrion et al. (2012) or Slade et al. (2009). Personal communication with the authors of Borrion et al. (2012) provided fossil fuel consumption rates for cellulase production. Slade et al. (2009) provide inventory data for one 'filter paper unit of cellulase'; however the conversion to 'kg enzyme' is not clear. Comparing the consumption of cellulase in the conversion process leads to an estimate of 2.2 to 3.8 kg CO_2 eq. tonne straw^{-1} (See **Appendix 6**). Due to a lack of data, these are assumed to be the same for wheat straw and Miscanthus.

Table 7-14. Chemical requirements for 2nd generation bioethanol.

Chemicals	Quantity (kg tonne biomass^{-1})	GHG Emissions (kg CO_2 eq. tonne biomass^{-1})
Enzymes/Yeasts	65.4[a]	147.3
Sulphur Dioxide (SO_2)	15.5[b]	6.7
Sulphuric Acid (H_2SO_4)	31.5[a]	3.9
Sodium Hydroxide (NaOH 50%)	29.0[b]	32.8
Ammonia (NH_3 25%)	2.4[a]	0.2
Ammonium Sulphate (NH_4)2PO_4	2.8[a]	7.5
Defoamer[c]	0.6[a]	-
Lime ($CaCO_3$)	23.0[b]	0.1
Diammonium phosphate	1.0[b]	0.3
Nutrient Feed[d]	13.1[b]	5.4
Total		206 -208

a) Slade et al. (2009)*Slade*

b) Borrion et al. (2012)

c) *Assume negligible*

d) *Assume rapeseed oil (A. Borrion pers. com. 2012)*

7.8.3. Energy requirements

Transparency in the LCA studies is limited with regard to heat and power requirements in the lignocellulosic ethanol plant. The NREL provide steam requirements for individual steps of the conversion process depending on various combinations of technological options. A process-flow model was developed by Borrion et al. (2012), adapting the processes for wheat straw. A heat and electricity requirement of 1.78 and 1.03 GJ tonne biomass^{-1} was calculated based on the model developed here. No information is provided on how much energy yields vary for

specific biomass types. The ranges given in **Table 7-15** are dependent on the bioethanol yields per tonne biomass.

There are conflicting reports for the energy requirements of 2nd generation lignocellulosic bioethanol production. In some studies it is assumed that the lignin production rate is sufficient to fulfil the heat and power requirements for the process (Aden et al., 2002; Wooley et al., 1999). Another study assumes that a natural gas requirement of 1336.7 MJ tonne biomass^{-1} is required for processing (Borrion et al., 2012b), and does not assume any exported electricity is produced. Another study assumes that lignin is not sufficient for electricity production and an import of 648 MJ tonne bioethanol^{-1} is required and that there is excess solid fuel (Slade et al., 2009).

This study has modelled a 2nd generation bioethanol plant with a CHP boiler. A lignin-specific boiler is not provided in the literature, therefore the CHP boiler model 3 is assumed to be appropriate. The power plant utilises the lignin stream for heat and power. If not sufficient then it is assumed to be supplemented by natural gas. If the lignin is produced in excess of power requirements then it is sold as excess solid fuel.

The lignin yield from 1 tonne of biomass is assumed to be 447 kg tonne straw^{-1} at 50%-63% moisture content (Borrion et al., 2012b). It is assumed to have a LHV of 6.68 to 9.89 GJ tonne^{-1}. The bioethanol plant has a total output of 235,233 tonnes bioethanol year^{-1} (Borrion et al., 2012b). The plant is assumed to be twice the size of a 1st generation bioethanol plant.

Table 7-15. Heat and power requirements for 2nd generation bioethanol (per tonne bioethanol).

Biomass	Biomass requirement (tonne)		Heat Requirement (MJ)		Power Requirement (MJ)	
	Min	Max	Min	Max	Min	Max
Wheat Straw	3.2	5.4	5633	9622	3267	5581
Miscanthus - Autumn	4.1	7.0	7295	12448	4231	7220
Miscanthus- Spring	3.7	6.3	6520	11125	3782	6453

7.8.4. Outputs from 2nd Generation Bioethanol Production

This section describes the outputs from 2nd generation bioethanol. These are identified as:

- Bioethanol
- Lignin
- Electricity
- Renewable chemicals
- Sludge (waste)
- Gypsum

The outputs of lignin, electricity and other co-products are discussed here.

7.8.4.1. Lignin

Lignin yields are generally not stated in the literature, as it is assumed that it is combusted onsite to provide heat and power demands. This study bases lignin production on Borrion et al. (2012), who estimated a yield of 447 kg tonne straw^{-1} at 63%-50% moisture content. Slade et al. (2009) estimated that after fulfilling the heat demands of the plant, an excess yield of 252 kg dry lignin tonne straw^{-1} is available to be sold.

7.8.4.2. Electricity

Electricity yields from lignocellulosic ethanol plants are highly uncertain (Wang et al. 2012). The NREL reports (Aden et al., 2002; Wooley et al., 1999) estimate an excess electricity generation of 0.4-0.6 GJ tonne biomass^{-1}. Slade et al. (2009) estimates a small import of 648 MJ tonne^{-1} bioethanol. In this study, based on the CHP unit specified, there is a net output of electricity of 2.4 to 5.0 GJ tonne bioethanol^{-1}.

7.8.4.3. Other Co-Products

The renewable chemicals that can be derived from lignin are not included in this assessment due to a lack of data on their relative yields.

7.8.5. Summary: Outputs from 2nd Generation Bioethanol

The main outputs from 2nd generation bioethanol are: bioethanol and exported electricity. **Table 7-16** summarises the quantities involved per tonne of wheat straw and Miscanthus input.

Bioethanol yields are a function of the biochemical composition of the material. One a per-tonne basis, spring-harvested Miscanthus yields the greatest amount of bioethanol. Therefore, production of one tonne of bioethanol would require between 0.2 and 2.1 hectares of arable land for wheat straw, 0.1 and 1.5 for Miscanthus if harvested in the autumn or 0.2 and 1.2 if harvested in the spring. This suggests that Miscanthus has the highest bioethanol yield potential.

Table 7-16. Summary of outputs from 2nd generation bioethanol production.

Biomass and Parameter	Value	
	Minimum	Maximum
Bioethanol Yield (tonnes tonne^{-1})		
Wheat straw	0.18	0.32
Miscanthus (autumn)	0.14	0.24
Miscanthus (spring)	0.16	0.27
Excess Electricity Output (GJ tonne^{-1})		
Wheat straw	2.42	4.13
Miscanthus (autumn)	3.13	5.35
Miscanthus (spring)	2.80	4.78

7.9. Summary: Bioethanol Production from Biomass

Bioethanol can be produced from wheat grain, wheat straw and Miscanthus. Conversion of wheat grain to bioethanol follows a modified version of conventional alcohol production, except there is an additional process of hydrolysis of starch using enzymes. The conversion of lignocellulosic material requires a higher level of processing due to the natural recalcitrance of the biomass. The process requires an acid pre-treatment phase which helps to degrade the biomass and make the 6 and 5 carbon sugars available for fermentation.

Processing the biomass requires both heat and power, which is best supplied through an inbuilt combined heat and power (CHP) plant. The plant would be scaled to provide the heat demands for the conversion process. Due to the higher processing requirements of 2nd generation biofuels, a larger CHP plant is required. In both 1st and 2nd generation bioethanol production case studies there is the option to utilise a renewable source of fuel for the CHP plant: from straw in the 1st generation scenario, and from the lignin waste stream in the 2nd generation example. Using lignin for onsite power generation should have the effect of reducing the GHG emissions from the overall process as it is assumed that biomass-based heat and power will be less GHG intensive as those from natural gas. In both cases, there is some exported electricity. Other co-products include DDGS and lignin; the former can be either sold for animal feed, or combusted for energy, and the upstream GHG emissions could be allocated by price, energy content or substitution credits awarded.

Transparent data on bioethanol is lacking, and there are few 1st generation and virtually no 2nd generation plants in existence, though data resources are limited. Therefore, the variation seen in the results will be assumed to be representative of general uncertainty in the inputs and outputs of bioethanol production.

Overall, there is uncertainty and variation in:

- Bioethanol yields
- Heat and power requirements for processing
- Exported electricity amounts
- Chemical requirements
- Substitution credits for DDGS
- Substitution credits for electricity (marginal or average)

The following Chapter examines the GHG emission results from the three ethanol routes and examines the sensitivity of the result to variability and uncertainty.

Chapter 8. Assessment of Variability

This chapter examines the GHG emission results from the two crop case studies. The results will evaluate the impact that variability and uncertainty in the collected data has on the calculated GHG emissions. This will be examined on a 'per hectare' basis for wheat so to avoid the impacts of LCA methodology for allocation between wheat grain and straw. For Miscanthus the results are examined on a per tonne basis, as there is temporal uncertainty that needs to be observed. The bioethanol production systems will show the GHG emissions from the whole process, also avoiding examining any impacts of LCA methodology at this stage of the study.

An analysis of variability and uncertainty is performed in order to examine causes of variation in LCA emission results due to 'real' differences in input data, and due to 'uncertainty' in the emissions that occur due to inputs or processes (Whitaker et al., 2010).

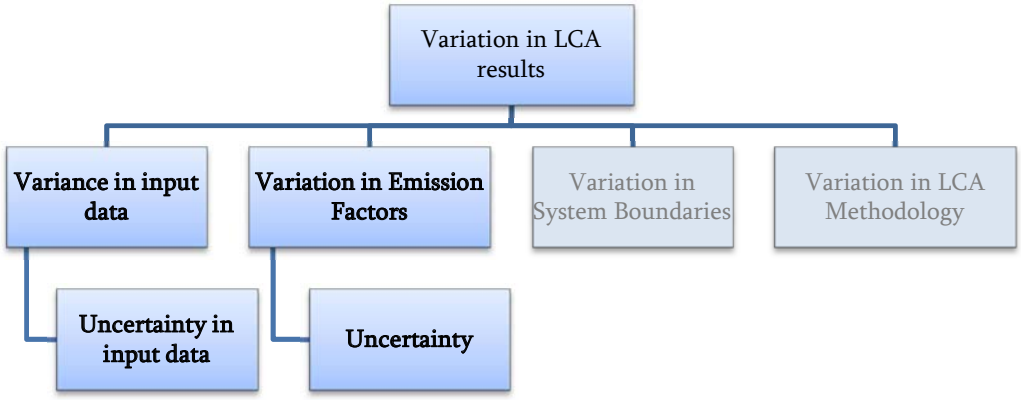

Figure 8-1. Focus of Chapter 9.

This chapter first examines variable and uncertain data sources in the case studies and then considers how these should be examined in a sensitivity analysis. The analysis includes variability and uncertainty in input data and emission factors (**Figure 8-1**).

This chapter specifically focuses on examining the impacts of:

- **Wheat**
  - GHG emissions from the farm sample
  - Total GHG emissions for an 'average hectare of wheat'
  - Sensitivity analyses:
    - Wheat variety
    - Input parameters
    - Direct land use change
    - Yield
    - The assumed probability density function
  - Relative effects of variability and uncertainty on the GHG emission results

- **Miscanthus**
    - GHG emissions from rhizome cultivation
    - GHG emissions from 'Industry standard' Miscanthus compared to literature
    - Autumn vs. spring harvesting
    - Artificial vs. organic fertilisers
    - Sensitivity analyses:
        - Input parameters
        - Direct land use change
        - Rotation length
        - The assumed probability density function
    - Relative impacts of variability and uncertainty

8.1. Variability and Uncertainty in the Collected Data

Chapter 5 provides an account of the inventory data collection phase. Variable inputs are regarded as those that are inherent to the system and cannot be reduced (Röös et al. 2010). The main type of uncertainty could be classified as 'epistemological' which is due to lack of knowledge of a system (Ahlgren et al., 2012). In some instances, for example when the collected data is based on theoretical data, it is not possible to identify whether variation is due to variability in inputs or due to uncertainty.

Attempts are made to identify whether ranges in the collected inventory data are due to variability or uncertainty. These are presented in **Table 8-1**, though may vary depending on the data that is available. The following section details how the ranges of the results are examined as part of an "uncertainty" analysis. Although it is termed as such, it is still possible to examine the effect of both variability and uncertainty on the results via an uncertainty analysis.

Table 8-1. Variable and uncertain parameters in wheat and Miscanthus cultivation.

Crop	Variable Aspects	Uncertain Aspects
Wheat	• Fertiliser rates • Machine operations • Pesticide use • Seed use • Yields	• Diesel consumption • Fertiliser penalty from straw removal • DLUC • N_2O emissions from soil
Miscanthus	• Yields • Fuel consumption rates • Fertiliser rates	• Fertiliser application rates • Pesticide application rates • Rotation period • DLUC

8.2. Uncertainty Analysis

Chapter 3 states that Monte Carlo analyses are identified in the literature as the preferred type of uncertainty analysis (Björklund, 2002). In this study, the methodology described in Guo & Murphy (2012) is followed. The analysis involves estimating the probability density function (*pdf*) of each parameter. The methodology is set out in **Appendix 7**. The *pdf* is deduced from data collected from growers, statistics, growers' guides and literature. Only in the farmer data, where there were many data points available, was it possible to analyse the data in this way. In the majority of cases the data is collected from separately estimated sources. For such "uncertain data" the minimum and maximum estimates are deduced and it is assumed that there is an equal, or uniform chance, of the parameter falling between the two extremes (Goedkoop et al., 2010).

In the literature, there are some estimates of the *pdf* for various parameters of arable and biofuel LCA studies. For example, Wang et al. (2012) estimates that fertiliser use and diesel use in energy grasses are normally distributed. Brinkman et al. (2005) estimates that input data for bioethanol yield and plant energy use is triangularly and normally distributed, respectively. Unfortunately to use the *pdf* information, one must have a substantial dataset in order to calculate a representative average and standard deviation. As the collected data is limited, these estimates cannot be used. Therefore in most cases, a uniform distribution is assumed. Ahlgren et al. (2012) also assumes a uniform distribution for artificial and organic N use, fertiliser manufacture and fuel consumption.

As the GHG emissions results are expected to be sensitive to fertiliser use and N_2O emissions from soil (Yan and Boies, 2013), the impact of the *pdf* on the GHG emission results will be examined for these parameters. **This will test the importance of understanding the distribution of likely results in the inventory dataset.** Here, comparisons will be made with N_2O emissions from soils following a triangular (Brinkman et al., 2005), uniform distribution (Ahlgren et al., 2012). Likewise, fertiliser application rates are assumed to have a triangular (this study for wheat), uniform (Ahlgren et al., 2012) or normal distribution (Wang et al. 2012).

8.3. GHG Emission Results in the Wheat Case Study

In the wheat case study, inventory data may vary in:

- **Fertilisers use**
- **N_2O Emissions from soils**
- **Diesel fuel consumption**
- **Pesticides**
- **Seed application**
- **Direct land use change effects**

The relative impacts of these parameters are discussed in the following sub-sections.

8.3.1. GHG Emissions from the Farm Sample

The GHG emissions from the farm sample are shown in **Figure 8-2**, with interquartile range shown with the error bars at 5% and 95% percentiles. The results are based on the data provided in the farmer questionnaires and does not include implications of DLUC or fertiliser penalties from straw removal. The range of the results represents uncertainty of the GHG emissions from soil, fertiliser manufacture and diesel consumption. This could be considered to provide a snapshot of the current 'industry' GHG emissions; however is mainly applicable to milling wheat.

On first inspection, it appears that there are one or two farms with a markedly higher GHG emission result (highlighted in red). Referring back to the inventory data unveils that these farms have received high quantities of organic fertiliser application. The high GHG emissions are associated with transporting them to the farm site and applying them. After discussing these results with an industrial advisor, it was decided to remove these farms from the sample as they do not reflect common practice, and in one case an application of coffee grounds was experimental. The farmers found that the high quantities of coffee grinds actually had a negative effect on the wheat yield. The farmer suggested it could be due to N immobilisation due to the higher C:N ratio of the coffee grounds (M. Carver pers. com. 2012). These farms are removed from the study. Please note that this does not exclude all farms that applied organic fertilisers, only those two that applied large quantities.

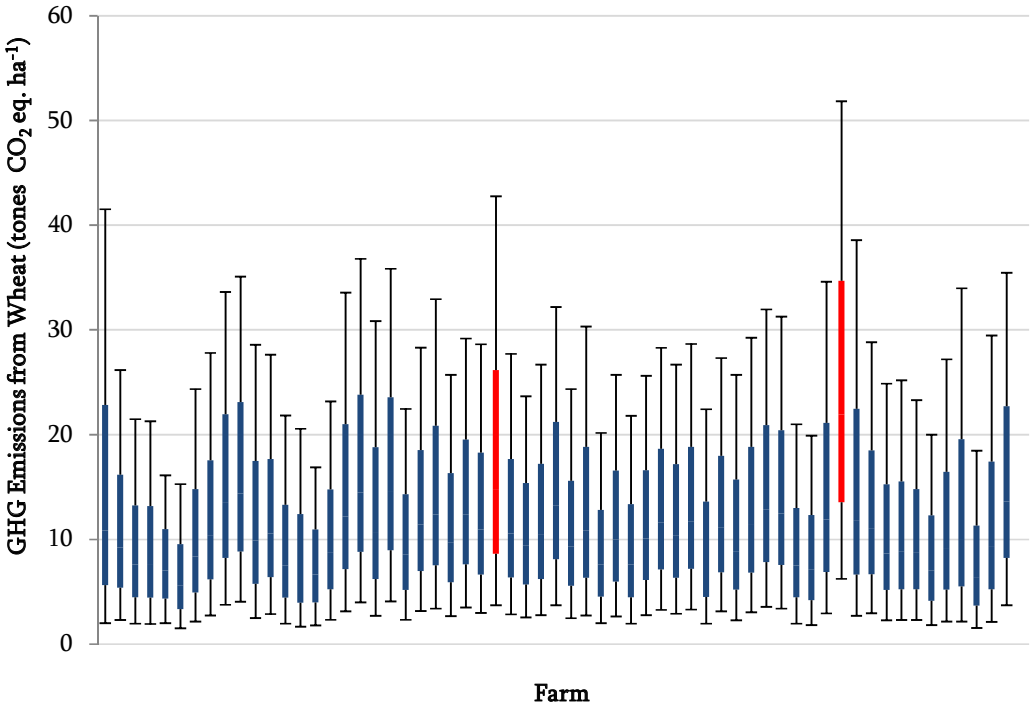

Figure 8-2. Array of GHG emission results from the 61 farm samples. Farms receiving unusually high quantities of organic fertiliser are in red.

8.3.2. Total GHG Emissions from Wheat: An 'Average Hectare'

This section discusses the GHG emissions from one hectare of wheat: **highlighting the most variable sources of GHG emissions on a per-hectare basis.** The main sources of GHG emissions from average wheat are shown in **Figure 9-3**. The results include all site inputs and exclude DLUC impacts. An average hectare of wheat in the UK releases 5,390 kg CO_2 eq. (128 kg CO_2 eq. ha^{-1} 95% CI), and there is evidence that this could range between 1,717 and 13,871 kg CO_2 eq. ha^{-1} due to variability in practices and uncertainty in inventory data. Fertiliser contributes, on average, 87% of total emissions.

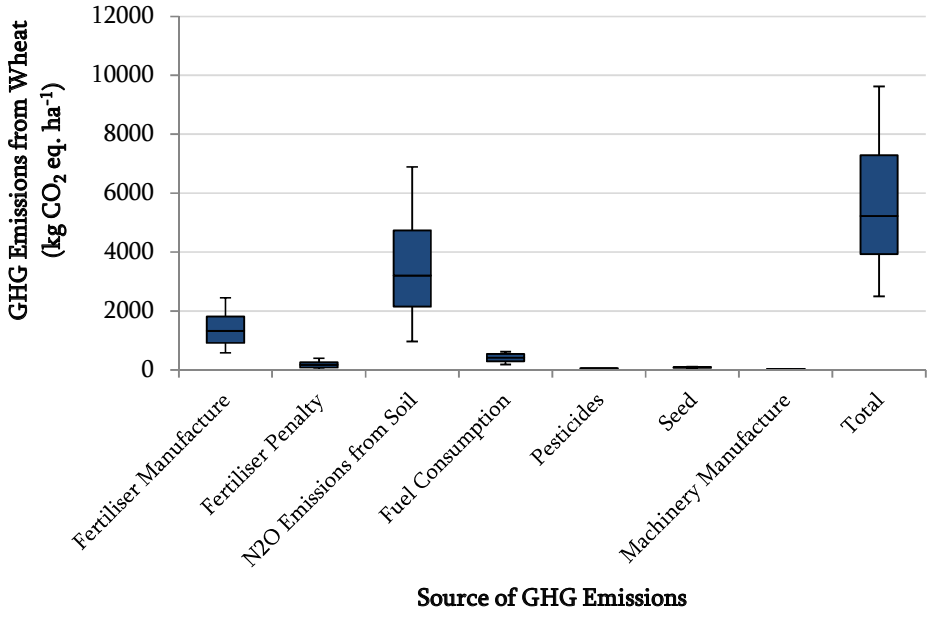

Figure 8-3. Summary of main sources of GHG emission's from wheat cultivation.

Emissions of N_2O from soil cause the majority of variation in the GHG emission results, followed by fertiliser manufacture. Fuel consumption for cultivation and drying contributes a modest proportion of total impacts (7%). Contributions from pesticides, seed and machinery manufacture are relatively small (**Figure 8-4**). The fertiliser penalty from straw removal also has a small impact on the results.

The GHG emissions from fertilisers originate from fertiliser manufacture and N_2O emissions from soil. The major sources of N_2O emissions from soil are identified as:

- **Direct due to N fertiliser application**
- **Indirect due to leaching and volatisation of applied N**
- **Direct due to crop residue incorporation**

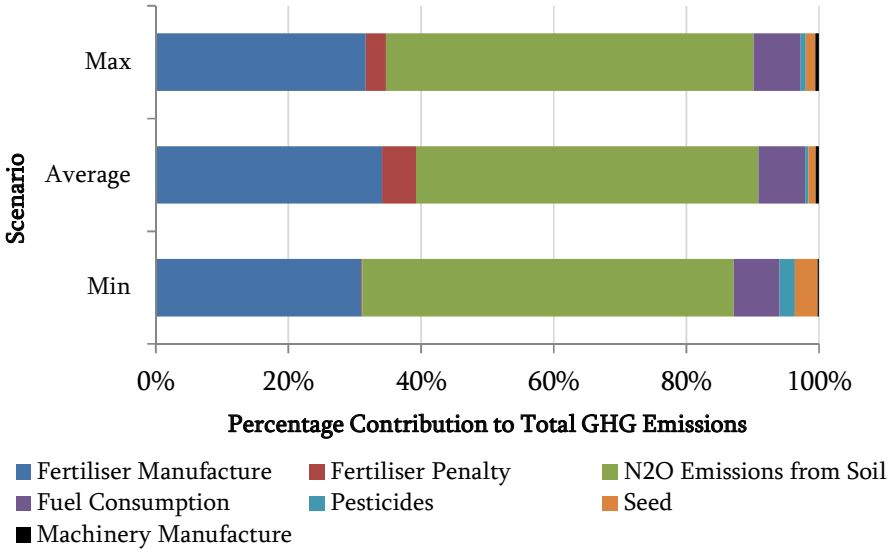

Figure 8-4. Relative contribution of each input to total GHG emissions, in minimum, average and maximum scenarios.

The majority of the N_2O emissions originate directly from N application to soils, and artificial fertiliser manufacture also makes a significant contribution (**Figure 8-5**). Overall, the GHG emissions from artificial fertiliser application are higher than for organic, mainly because of increased manufacturing GHG emissions. If fertiliser manufactured under best available technology (BAT) is used then this reduces the overall GHG emissions from artificial fertiliser application by 26%.

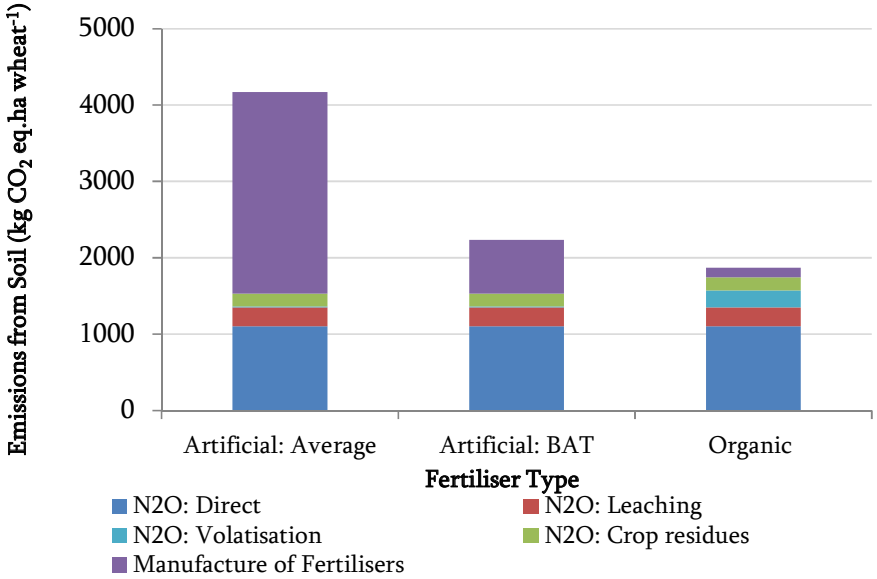

Figure 8-5. Sources of N_2O emissions from soil, based on average values and fertiliser type.

8.3.3. Sensitivity Analyses

The sensitivity of the final GHG emission results to specific parameters are examined here. The results include the sensitivity to the wheat variety, to the main inputs to wheat cultivation and to land use change (LUC) impacts. Finally the sensitivity of the GHG emission results according to assumptions of the *pdf* is assessed.

8.3.3.1. Sensitivity to Wheat Variety

Two varieties of wheat were examined in the wheat case studies: feed and milling wheat. Compared to feed wheat, milling wheat has a higher N fertiliser application rate of 40-60 kg ha^{-1} more N applications per hectare (Agro Business Consultants Ltd, 2011; MAFF, 2000; Nix, 2011a). **Figure 8-6** shows how the increased N fertiliser rates of milling wheat means that not only the GHG emission rates per hectare are higher, but they have a higher variation also. This is due to a combination of increased uncertainty in fertiliser manufacture GHG emissions and N_2O emission rates from soil. Differences in yields and economic values of the two varieties will also cause differences during allocation.

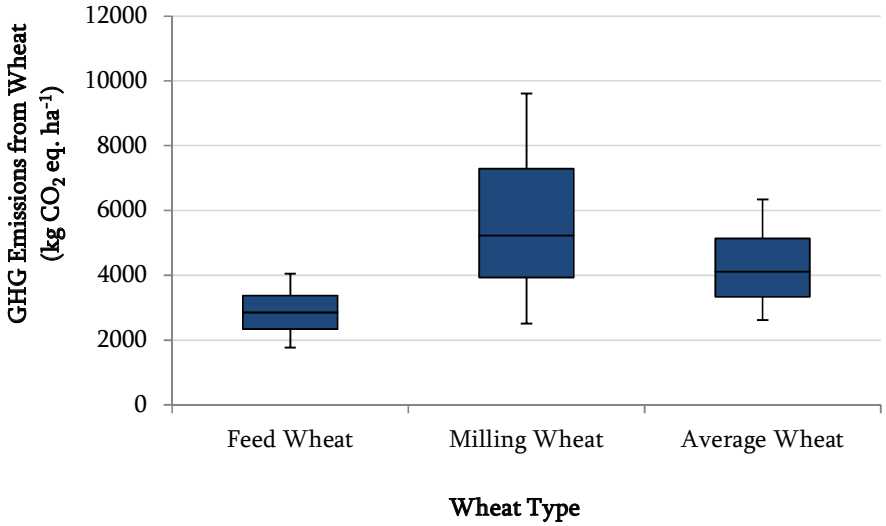

Figure 8-6. Wheat variety and GHG emissions.

8.3.3.2. Sensitivity to Parameters

As described in Chapter 3, a sensitivity analysis will indicate which parameters the results are most sensitive to. A sensitivity analysis provides some indication of the influence of the most important assumptions in a LCA study (Goedkoop et al., 2010). This is performed by changing the values of yields, fertiliser application rates, etc., by continuous decrements and increments from minus to plus 100%. The resulting GHG emission result is recorded and a linear regression is plotted against each parameter and the gradients (m) ranked to identify the order of sensitivity.

Figure 8-7 shows the results of the analysis for the input-based parameters, including fertiliser, fuel consumption, machinery use, pesticide and seed use. It shows that the GHG emissions are highly sensitive to the N_2O emissions rate from soil and the N application rate to soil. As discussed in Section 8.4.2, GHG emissions from fertiliser manufacture and N_2O emissions contribute 87% of total GHG emissions from wheat cultivation. The results show however, that the GHG emissions are also highly sensitive to these parameters.

Other parameters show less importance in terms of overall sensitivity in the results. These parameters include organic fertiliser, P_2O_5 and K_2O-based fertilisers, drying, seeds, machinery manufacture and the fertiliser penalty from straw removal. The results show a higher sensitivity to the N-based fertiliser penalty for straw removal compared to P_2O_5 and K_2O-based fertilisers, indicating the importance in understanding the N-based implications of straw removal. When the sources of N-based fertiliser are separated (not shown) it is apparent that the results are most sensitive to application rates of ammonium nitrate.

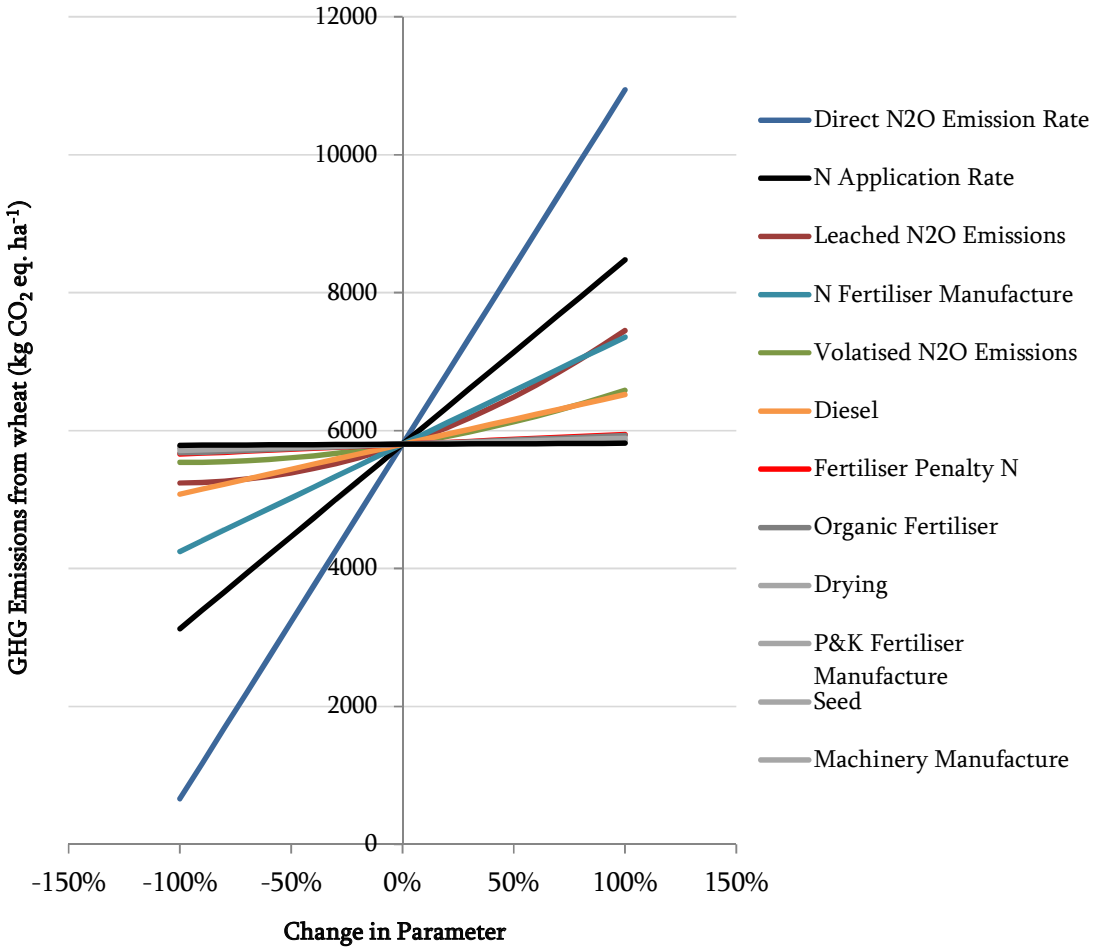

Figure 8-7. Sensitivity of GHG emissions from wheat according to input parameters.

8.3.3.3. Sensitivity to Land Use Change Impacts

In wheat, relevant land use change events involve conversion of forestland or grassland to arable land, as well as changes in residue management. These must be accounted for if land conversion has taken place within the previous 20 years (Bickel et al., 2006). The effects of these on the GHG emission results are shown in **Figure 8-8**. The results show that conversion of forestland or grassland for arable crop production can have a very large impact on the GHG emissions from wheat cultivation. In all cases, DLUC is expected to cause losses of SOC. This is seen as an increase in GHG emissions per hectare of wheat. Conversion of forestland incurs the greatest net GHG emissions, followed by conversion of grassland. The impacts of removal of straw increases the GHG emissions per tonne by at least 20% compared to when DLUC is excluded in the accounts.

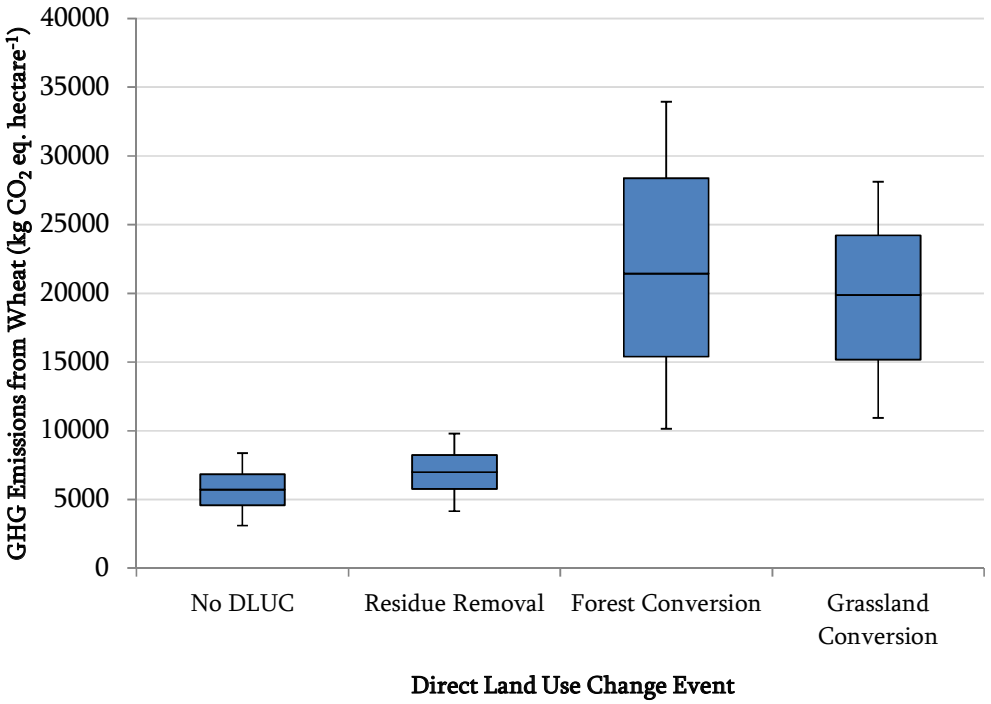

Figure 8-8. Impacts of DLUC events on the GHG emissions from wheat.

8.3.3.4. Sensitivity to Yield

Although the previous analyses avoid examining the impact of allocation between wheat grain and wheat straw by only examining the results on a per-hectare basis, it is examined here as part of a sensitivity analysis. Allocation is performed according to price, mass and energy content. The results are based on the net output of economic value and energy yield from a hectare of wheat (**Figure 6-9** in **Chapter 6**). This assumes that wheat grain incurs 57%-96% of the GHG emissions from wheat cultivation, and wheat straw takes the remainder. The GHG emissions 'per tonne' then depend on the yields of wheat and straw.

Figure 8-9 shows the proportion of GHG emissions allocated to wheat grain and wheat straw. The upper and lower limits are a combination of the upper and lower GHG emissions calculated due to variability and uncertainty, as well as the upper and lower allocation proportions.

When allocating by price wheat grain has a higher average GHG emission than for wheat straw. When allocated by mass and by energy content wheat straw has a higher overall GHG emission per tonne of grain (**Table 9-2**). Straw incurs an additional 20 kg CO_2 eq. tonne^{-1} on average from a separate baling process, and 178 kg CO_2 eq. tonne^{-1} to replace nutrient off-take.

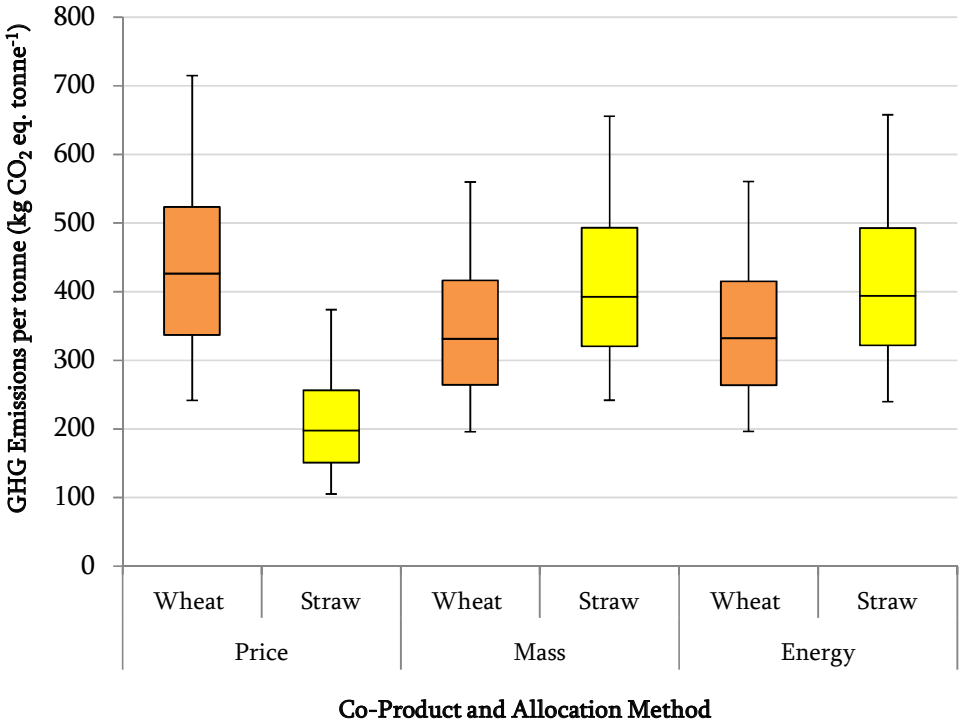

Figure 8-9. GHG emissions between wheat grain and straw according to allocation method (based on average wheat)

Table 8-2. Example of allocation between wheat grain and straw by mass and by price.

Allocation by	Total GHG Emissions (kg CO_2)	Allocation (%)		Yield (tonnes ha^{-1})		Additional (kg CO_2 eq. tonne^{-1})	Total GHG Emissions (kg CO_2 eq. tonne^{-1})	
	Per ha	Wheat	Straw	Wheat	Straw	Straw	**Wheat**	**Straw**
Mass	3227	65%	35%	10	5.0	64	**210**	**290**
Price		86%	14%				**278**	**154**

The range of the GHG emission factor per tonne of wheat or straw varies the most when allocated by price, perhaps due to temporal variation seen in wheat grain prices. Straw has a large range in emissions as there is uncertainty in the fertiliser penalty, which is entirely attributed to straw. The difference is negligible between the allocation proportions for wheat and grain with mass and energy content. Wheat grain and wheat straw have similar energy contents (see **Table 6-13** of **Chapter 6**).

After first assessing the difference that alternative allocation methods make to the GHG emissions per tonne, the sensitivity to yield is assessed. **Figure 8-10** shows the results of the sensitivity analysis for the GHG emissions of wheat grain. The results for wheat straw would be mirrored as the GHG emissions are shared between the two co-products. The GHG emissions per tonne of wheat grain are highly sensitive to the yield and price grain. As the inherent price of grain is so high the GHG emissions are not as sensitive to the price or yield of straw. The energy content of wheat and straw is similar therefore these two co-products have equal sensitivity.

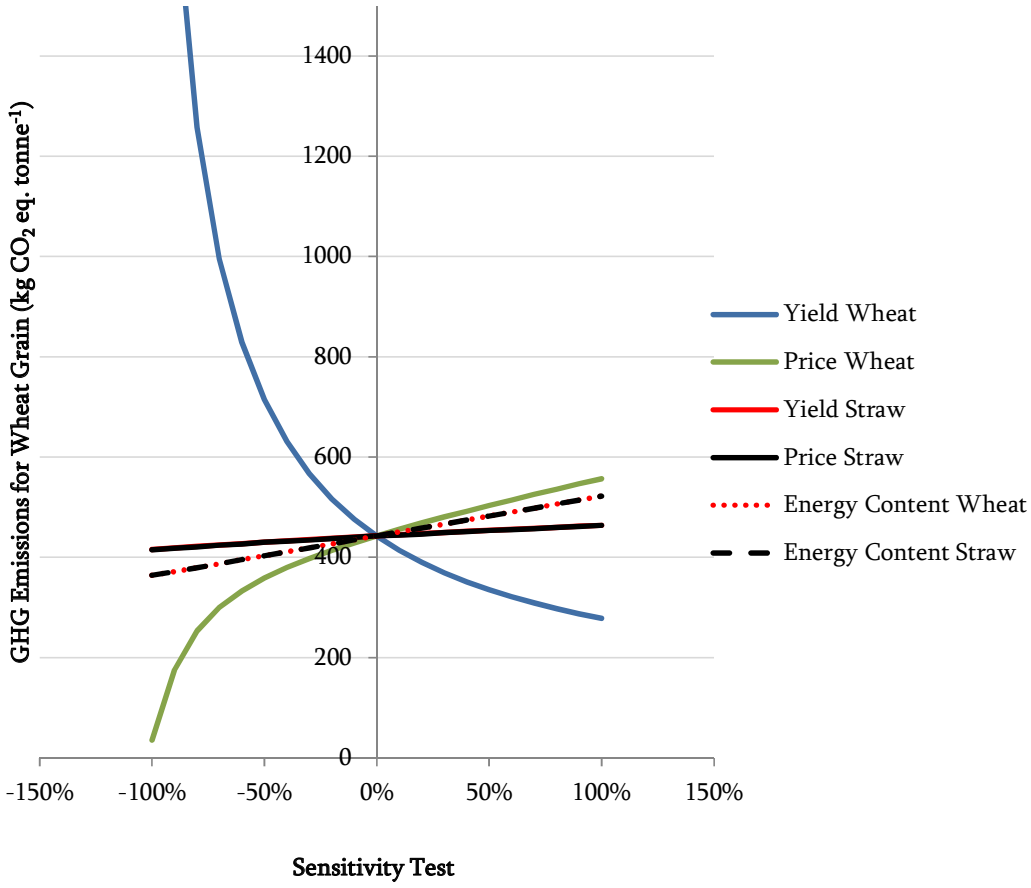

Figure 8-10. Sensitivity analysis for the GHG emissions per tonne of wheat grain.

8.3.3.5. Sensitivity to the Probability Density Function

In the study, a triangular distribution was assumed for N-based fertiliser application rates to soil and a uniform distribution is assumed for all other parameters. Assuming a triangular distribution involves assuming a minimum and maximum point and a "best guess" value. A uniform distribution only considers points between the minimum and maximum points and assumes there is an equal chance of the parameter falling between these two points.

The per-hectare GHG emission results are highly sensitive to N application to soil as this is directly associated with direct and indirect N_2O emissions from soil and also GHG's from manufacture of N-based fertilisers. Therefore a sensitivity analysis of assumptions of the probability density function (*pdf*) of N application rates is performed. As described in Section 8.3, this involves comparing the N_2O emission rate from soils following a triangular (Brinkman et al., 2005), uniform distribution (Ahlgren et al., 2012) and fertiliser application rates with a triangular (this study for wheat), uniform (Ahlgren et al., 2012) or normal distribution (Wang et al. 2012).

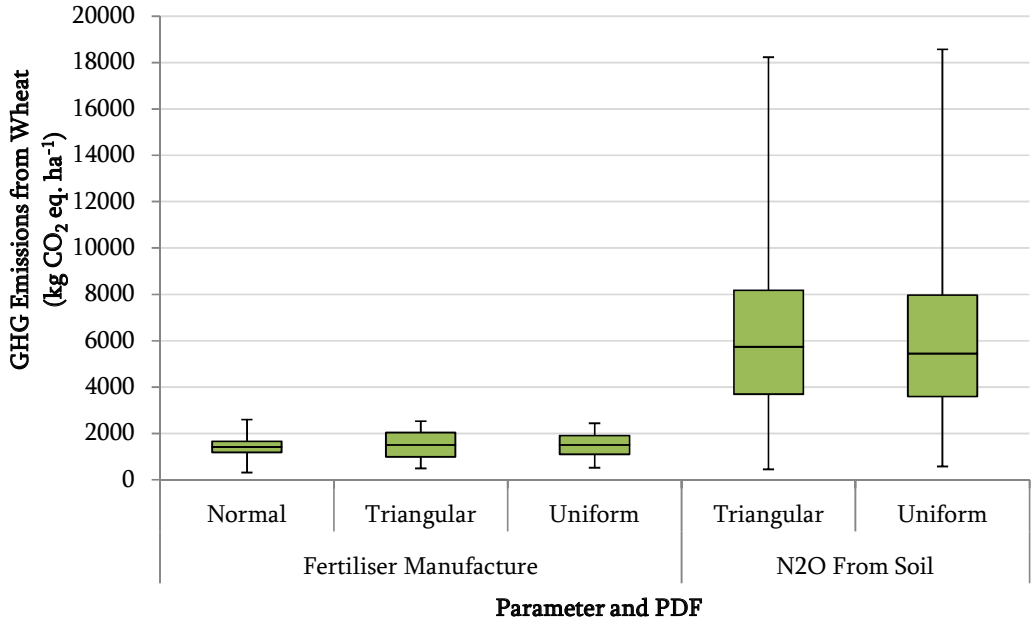

Figure 8-11. Effect of *pdf* assumptions on the GHG emission results.

It can be seen in the results that assumptions on the *pdf* have a relatively small impact on the range of results (**Figure 8-11**). The largest differences appear to be in the width of the interquartile range rather than the overall range. **It can therefore be deduced from this sensitivity analysis that the effect of the *pdf* on the results is relatively small.**

8.3.4. Relative Impacts of Uncertainty and Variability

This section examines how the GHG emissions from wheat vary due to variation or uncertainty. This will highlight the cause of the variation of the GHG emission results and determine where such variation can or cannot be reduced by increasing knowledge of the system.

The wheat case study described in **Chapter 6** identified the following sources of variation and uncertainty. These will be examined in a series of scenario runs. These are performed by 'freezing' variation of all other parameters and isolating variation in a particular parameter.

- Variation
  - Spatial variation (Run 1) – variation in site inputs, yields, as recorded in farmer data
  - Temporal variation (Run 2) – variation in yields over last 10 years
- Uncertainty
  - Variety of wheat (Run 3) – feed wheat, milling wheat or 'average wheat'
  - Manufacture of inputs (Run 4) – fertilisers
  - DLUC due to Changes in residue management (Run 5)
  - Fuel Consumption (Run 6)
  - N_2O emissions from soils (Run 7)

A further Run 8 will examine the combined effects of all uncertainty on the GHG emission results. **Figure 8-12** shows the results of these runs.

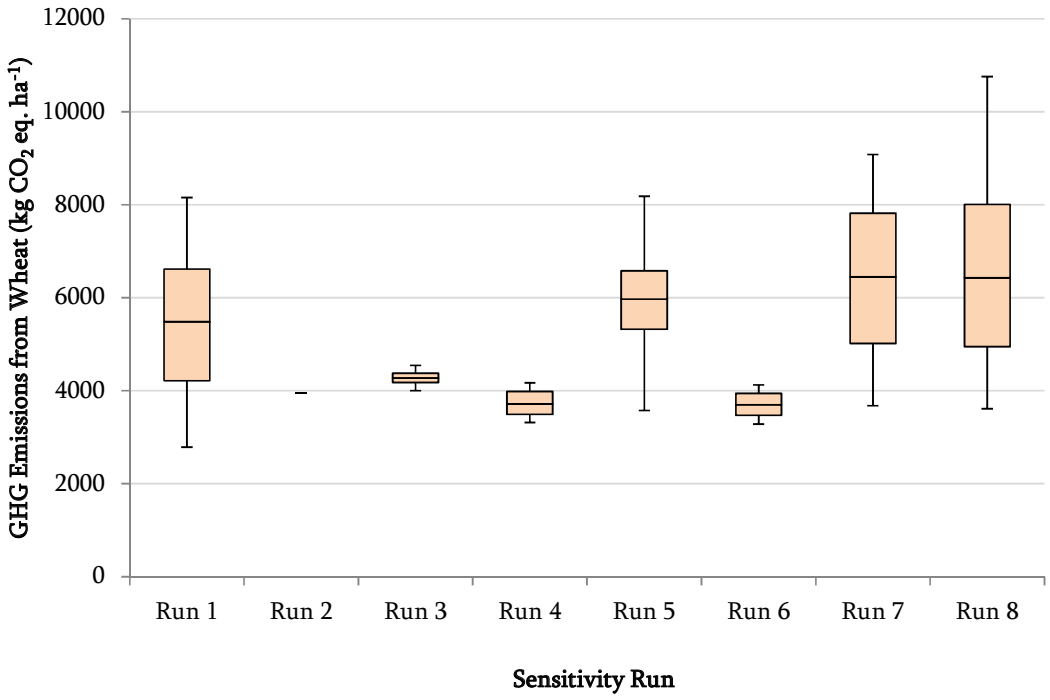

Figure 8-12. GHG emission results for wheat cultivation in Runs 1-8. This includes DLUC due to changes in residue management.

From the array of runs, it appears that numbers 1, 5 and 7 causes the majority of the variation in Run 8, which is the total variation in the GHG emission results. Run 1 shows variability in the inputs seen across sites in the UK from specific farms, and the majority of the variation is due to N-based fertilisers.

There is a great deal of uncertainty in the effects of DLUC on GHG emissions from land. As shown in the figure 'Run 5' represents the DLUC implications of changes in residue management. There is also a great deal of uncertainty associated with conversion of forest or grasslands to arable land. These are regarded as "uncertain" as the extents of changes in SOC are currently based on literature estimates. The results show that there is a large range in N_2O emissions from soils, which are seen in Run 7. Such variation is due to both ranges in N applied across sites and the range in emission factors for applied N, however **Figure 8-12** shows that there is greater variation in the latter due to uncertainty in emission factors for N application to soil.

8.3.5. Summary: GHG Emissions from Wheat

In this study, it can be concluded the GHG emissions from a hectare of wheat is highly sensitive to the N application rate and grain yield. The nature of the fertiliser should be recorded as the GHG emissions from manufacture will vary considerably between fertiliser types, particularly if manufactured under BAT practices or derived from an organic source. These results are comparable to those in other studies, such as Ahlgren et al. (2012), Rowe et al. (2011) and Yan & Boies (2013).

The analyses of uncertainty, variability and sensitivity performed in this chapter on wheat cultivation have identified the following major and minor sources of variability in the results:

- "Major" sources of variation
    - Fertiliser application rates
    - Fertiliser types applied
    - N_2O Emissions from soil
    - DLUC from changes in residue management, conversion of grass and forestland
- "Minor" sources of variation
    - Fuel consumption
    - Variety of wheat
    - Fertiliser penalties from straw removal
    - Seed use
    - P_2O_5 and K_2O-based fertilisers
    - N_2O emissions from stubble incorporation

8.4. The Miscanthus Case Study

In the Miscanthus case study, inventory data may vary in:

- **Fertilisers**
- **N_2O Emissions from soils**
- **Diesel fuel consumption**
- **Pesticides**
- **Direct land use change effects**

This section describes the GHG emissions from Miscanthus cultivation. First the results of rhizome propagation are presented. Then the total GHG emissions from main crop Miscanthus cultivation are shown on a per-tonne basis. The GHG emissions per hectare per year will depend on the yield, which depends on whether Miscanthus is harvested in the autumn or the spring. A series of sensitivity analyses are then performed.

8.4.1. Rhizome Propagation

This section discusses the GHG emissions from the rhizome propagation phase which involves site establishment, harvesting, sorting and storing the rhizomes. Overall, one hectare of rhizome cultivation, which may last between three and five years, causes the GHG emissions of 2,042 kg CO_2 eq. ha^{-1} (37 kg CO_2 eq. ha^{-1} 95% confidence interval (CI)), and ranging between 847 and 4,476 kg CO_2 eq. ha^{-1}.

As one hectare yields between 20 and 28 tonnes of rhizomes, one tonne of rhizomes releases 86 kg CO_2 eq. ha^{-1} (2kg CO_2 eq. $tonne^{-1}$ 95% CI), ranging between 49 and 131 kg CO_2 eq. $tonne^{-1}$. Based on a typical weight of rhizomes of between 20 and 50 grams, or 0.002 to 0.004 kg CO_2 eq. $rhizome^{-1}$. This is an order of magnitude lower than the figure of 0.015 kg CO_2 eq. $seedling^{-1}$ for in vitro propagation (Felten et al., 2013). Therefore it can be assumed that rhizome multiplication method is more efficient than *in vitro* propagation.

Figure 8-13 demonstrates the sources of GHG emissions during the production of rhizomes. Here it is assumed that the rhizomes are treated with artificial fertilisers on establishment: therefore is a 'worst case' scenario, as commercial growers do not apply fertilisers to these crops.

The results show that the GHG emissions are mainly dominated by fertiliser manufacture and application, though this would not be the case according commercial stands. The next important source of GHG emissions are from fuel consumption. Very minor GHG emissions occur from machinery manufacture, pesticides, sorting and storage.

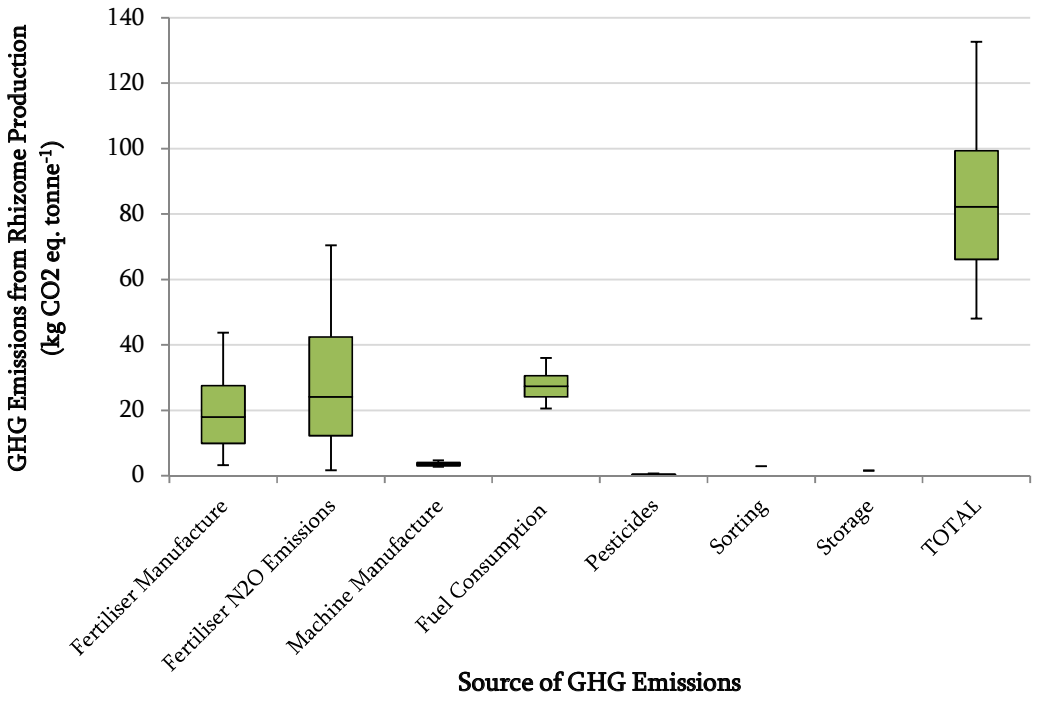

Figure 8-13. Sources of GHG emissions for rhizome cultivation according to literature.

8.4.2. Main Crop Miscanthus Cultivation

This section discusses the GHG emissions from Miscanthus cultivation. The analysis includes site establishment, first year maintenance, harvesting and termination. It is assumed that the crop has a productive life of between five and 20 years. The data is provided from both contractors and literature and includes estimated ranges of inputs of fertilisers, fuel and ranges of yields of Miscanthus biomass.

As the GHG emissions of the crop are examined in several ways, this section is split into the following sub-sections:

- **"Industry standard" Miscanthus vs. literature estimates**
- **Literature-based Analyses:**
  - **Autumn and spring harvesting**
  - **Artificial vs. organic fertilisers**

8.4.2.1. "Industry Standard" Miscanthus vs. Literature Estimates

Miscanthus cultivation data was collected from an industrial contact in the UK. Literature was also consulted to cross-reference with the data from contractors. An account of the collected data is provided in **Chapter 7**. The main differences in the two data sets include fertiliser use data and yield.

Industrial production of Miscanthus is reported to only require 40 kg K_2O ha^{-1} to be applied every five years, whereas there is a large range of reported fertiliser rates for Miscanthus in the literature. The yield of Miscanthus in the field (8.0 tonnes ha^{-1}) is lower than reported for UK-average in the literature (13.9 tonnes ha^{-1}).

Figure 8-14 compares the industrial standard of Miscanthus cultivation with data from literature. In some cases there are some overlaps where there are data gaps, such as fertiliser requirements after termination of the crop. **The results clearly show that 'industrial Miscanthus' has a much lower GHG emission profile than information in the literature.**

The results suggest this is solely a consequence of assumptions on fertiliser use in the crop. In the literature-based dataset fertiliser manufacture and N_2O emissions from soil contribute 86% of total GHG emissions. In the industry standard this is 13%.

Both the fertiliser use and the yield in the literature are higher. There is insufficient evidence, however, that this is due to a yield response to fertiliser. In either case, the results suggest that an increase of 4 tonnes ha^{-1} does not compensate for the use of fertilisers, as the GHG emissions are still higher than with the lower yielding commercial crop.

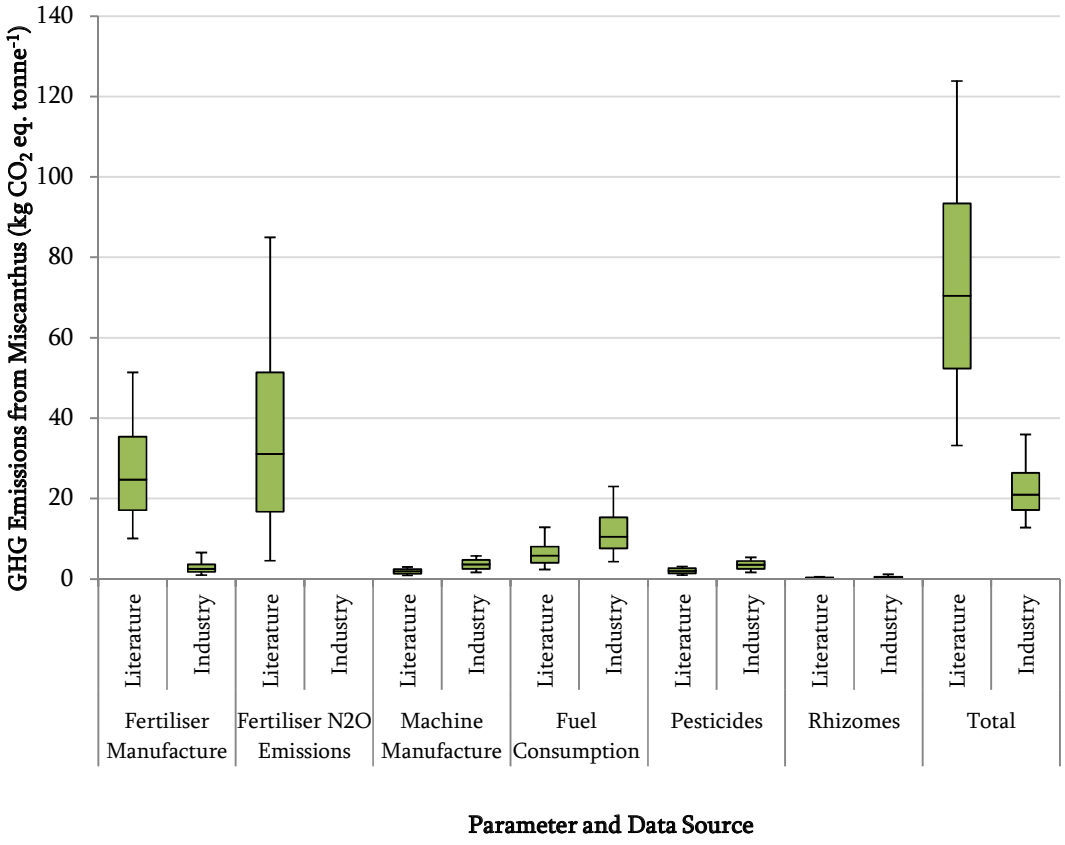

Figure 8-14. GHG emissions from Miscanthus cultivation based on Industrial and literature-based data.

The results demonstrate the importance of understanding the real nutrient demands of Miscanthus. It is arguable that the industrial standard of Miscanthus proves that the nutrient demands of Miscanthus are low, as these stands are currently in commercial use. They are, however, relatively young (under 10 years old), so the future nutrient demands may be yet to be fully realised. There may be some site-specific impacts, such as residual nutrients from the previous cropping cycles. Further research may be required to fully confirm and validate the results seen in the industrial crops.

In the literature-based data the GHG emissions from fuel consumption are relatively low (7%) however overall they represent 52% of the emissions from the industrial Miscanthus case study.

In both industrial and literature-based datasets, rhizomes represent a maximum of 1% of the total GHG emissions, or on average 0.2%. Likewise, the maximum contributions that machine manufacture and pesticides make are 3% each. If the 'material contribution' of GHG emissions is limited at 5%, these individual sources of GHG emissions would fall below the threshold.

8.4.2.2. Autumn vs. Spring Harvesting

The impacts of autumn harvesting and spring harvesting are compared here. It is assumed that autumn harvesting will incur a greater fertiliser requirement due to increased off-take of nutrients (Clifton-Brown et al., 2007). The data is based on literature data as autumn, or 'green harvesting', is not currently carried out in commercial crops as there is a demand for drier Miscanthus, and there is scepticism to the long-term success of harvesting early (M. Carver, pers. com., 2012). Nutrients are relocated from the above-ground biomass to the rhizomes for winter storage and if harvesting occurs before this point then it is assumed that the crop would become increasingly 'exhausted'. The results suggest that this may be avoided by increasing nutrient application rates in autumn harvested crops.

Benefits for harvesting in the autumn are that biomass yields are expected to be higher by an average of 6 tonnes ha^{-1} compared to the spring harvest. **Figure 8-15** compares the GHG emissions from producing one tonne of Miscanthus by autumn or spring harvesting. The overall ranges with minimum and maximum points are shown along with the interquartile range. It does not include DLUC impacts.

The results suggest that the GHG emissions from increased fertiliser application rates in autumn harvesting are not compensated by the higher yield. If autumn harvesting requires additional fertiliser application, then it may not be advantageous for the GHG emission balance of the crop. This is despite the crop producing a higher amount of biomass from the same amount of land.

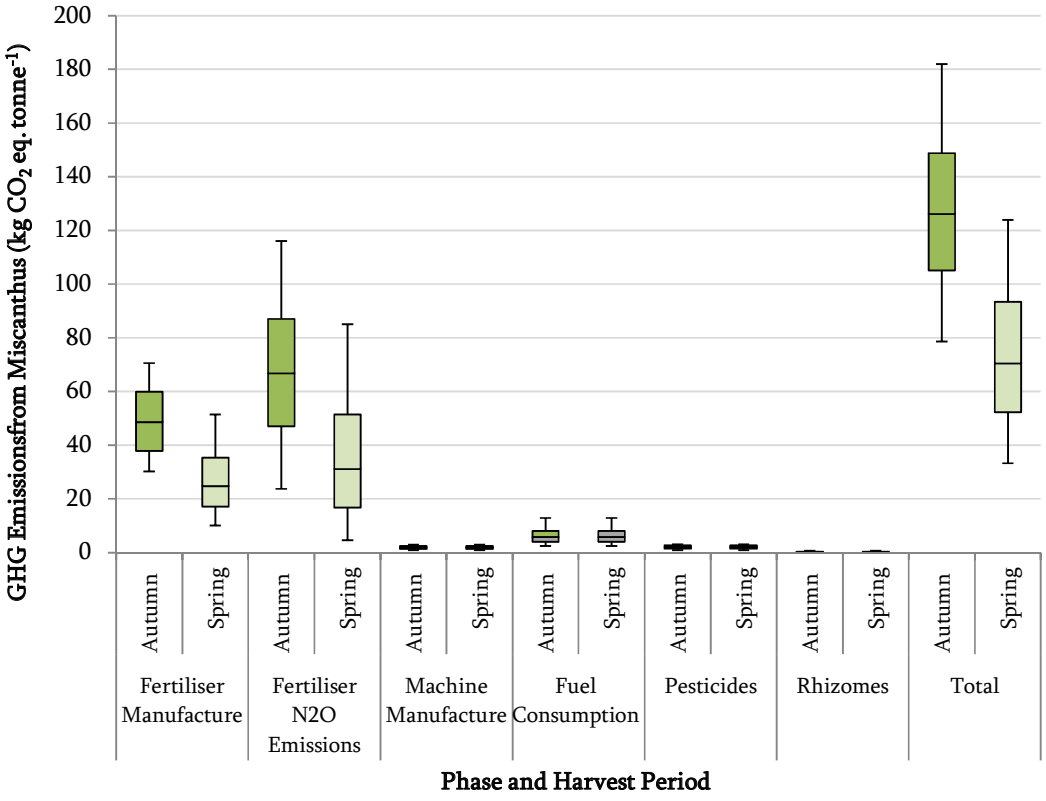

Figure 8-15. GHG emissions from Miscanthus cultivation according to cultivation phases and harvest time. This is based on literature data.

On average, the GHG emissions from Miscanthus cultivation are 68% higher if harvested in the autumn. If the increased fertiliser requirements are excluded, the higher yields in autumn mean the GHG emissions are 7% lower than spring harvesting. Therefore the results are less sensitive to yield compared to fertiliser applications. The GHG emissions from fertiliser requirements may be overestimated, as the contractors only recorded one application of K_2O after five years. Despite this, the model dataset shows that if autumn harvesting incurs a greater fertiliser requirement of just 40 kg N ha^{-1} then the GHG balance is negatively affected. Likewise, an application of 1 tonne ha^{-1} $year^{-1}$ of organic fertiliser will increase the overall emissions more than the yield reduces them.

8.4.2.3. Artificial vs. Organic Fertilisers

The impacts of utilising artificial or organic fertilisers during main crop Miscanthus cultivation are examined here. **Figure 8-16** shows the GHG emission results comparing the two fertiliser types. The study has not captured the difference that either fertiliser would make to GHG contributions from machine manufacture or fuel consumption. For example, organic fertiliser application to soil requires different machinery and will have a slower application rate (Nix, 2011a). This is not measured here as the difference is expected to be small.

The results suggest that organic fertilisers give a higher GHG emission rate compared to artificial if the same quantities of nutrients are applied to the site. This is an interesting result as artificial fertiliser are much more GHG-intensive to manufacture than for organic. The main issue here is the concentration, or availability of nutrients within organic fertilisers.

Miscanthus requires K_2O additions more than any other nutrient, yet this is found in the lowest concentrations in organic fertilisers. Therefore, to ensure an application of sufficient K_2O, there is an unavoidable excess application rate of N and P_2O_5. Artificial fertilisers can more easily control the quantities of nutrients applied to the crop. Less N is applied in the artificial fertiliser scenario; hence the GHG emissions are overly lower.

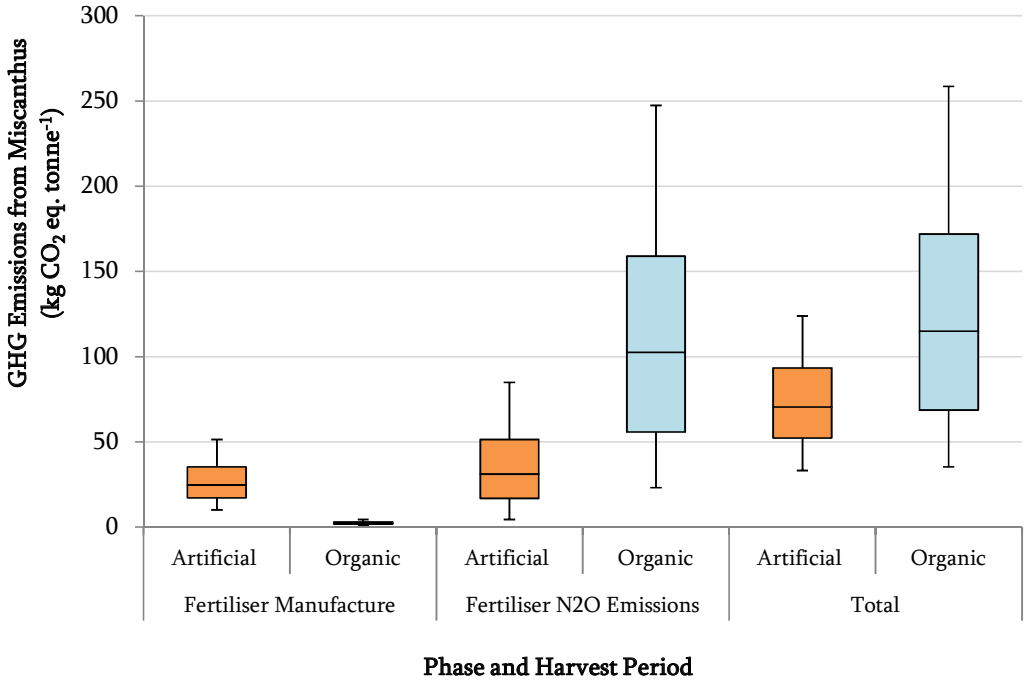

Figure 8-16. GHG emissions from Miscanthus cultivation when harvested in the spring and treated with either artificial or organic fertilisers.

8.4.3. Sensitivity Analyses

This sub-section examines the sensitivity of the final GHG emission results to specific parameters. First, the sensitivity of the results to the main inputs to Miscanthus cultivation is determined. The sensitivity to land use change (LUC) impacts are then examined, comparing the effects of direct LUC from land conversion of arable land, grassland and forestland to Miscanthus. Finally the sensitivity of the GHG emission results according to assumptions of the *pdf* is assessed.

8.4.3.1. Sensitivity to Parameters

As with the wheat cultivation case study, the sensitivity analysis is performed by changing the values of yields, fertiliser application rates, etc., by continuous decrements and increments from minus to plus 100%. The resulting GHG emission result is recorded and a linear regression is plotted against each parameter and the gradients (*m*) ranked to identify the order of sensitivity. **Figure 8-17** shows the results of this analysis for the input-based parameters.

There are some different aspects of sensitivity compared to those seen in the wheat case study. The results show a high sensitivity to annual yield. This is because each increase of 1 tonne affects every year the crop is harvested (on average 12 years), and increases the net output of biomass from the site. An interesting anomaly can be seen in the results with regard to rotation length. Shorter rotation periods have a tendency to increase the GHG emissions per tonne significantly; however there is a turning point after which this impact is levelled out.

Other parameters show less importance in terms of overall sensitivity in the results. These parameters include pesticides, indirect N_2O emissions, rhizomes, diesel, machinery manufacture and crop residue deposition.

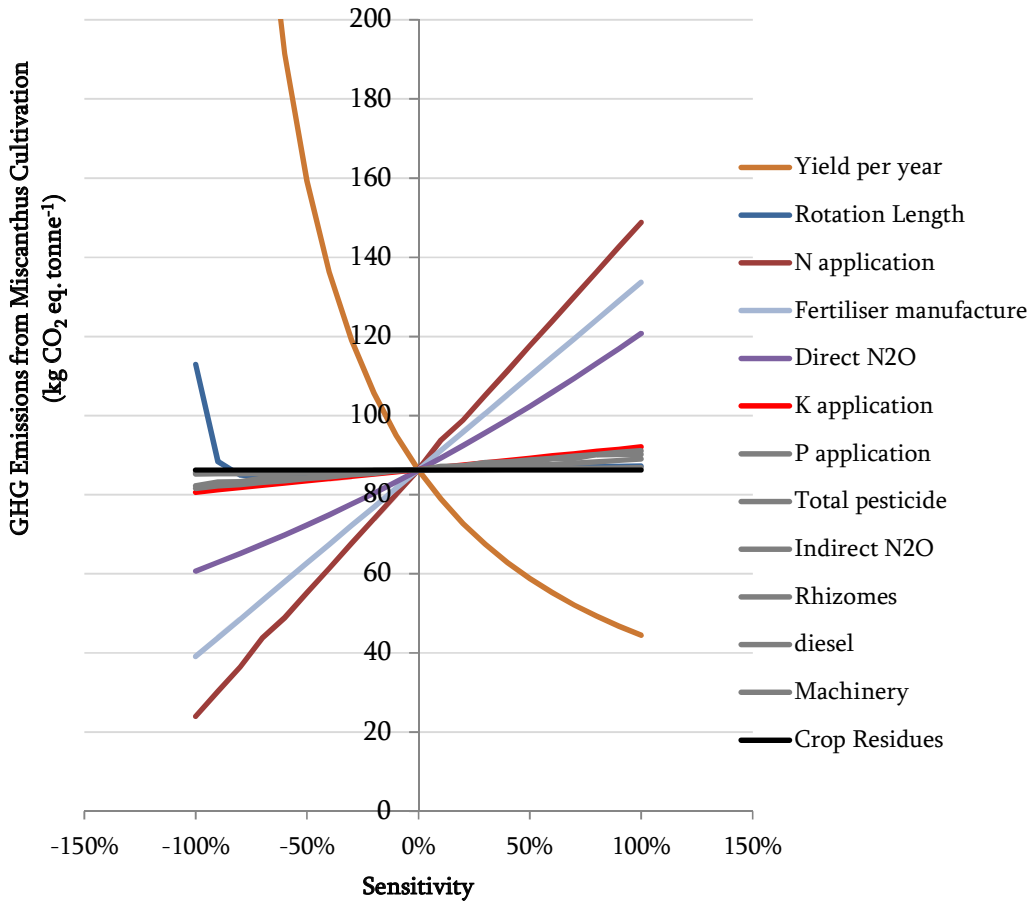

Figure 8-17. Sensitivity analysis of GHG emissions from Miscanthus.

8.4.3.2. Sensitivity to Land Use Change

There is evidence that autumn harvesting decreases the rate of carbon sequestration under the soil because less biomass is lost from the crop (in the form of leaves and debris) during the winter period (Amougou et al. 2011, Grogan & Matthews 2001). The impact this has is to increase the GHG emissions of harvesting Miscanthus in the autumn (**Figure 8-18**).

It can be seen that the RED and IPCC's estimate for carbon sequestered under Miscanthus is higher than the rate estimated after literature review. The difference may be because the two estimates have followed a different methodology in which to estimate SOC sequestration. **Therefore the RED and IPCC currently overestimate the carbon sequestration potential of Miscanthus.** The calculations here apply results from literature, which includes both modelled results and field trials. The RED and IPCC methodology is based on default numbers. Another cause of variation between the results may be because the RED's and IPCC's figures do not include SOC lost during the termination of the crop, which may reduce the total sequestered carbon by 34%.

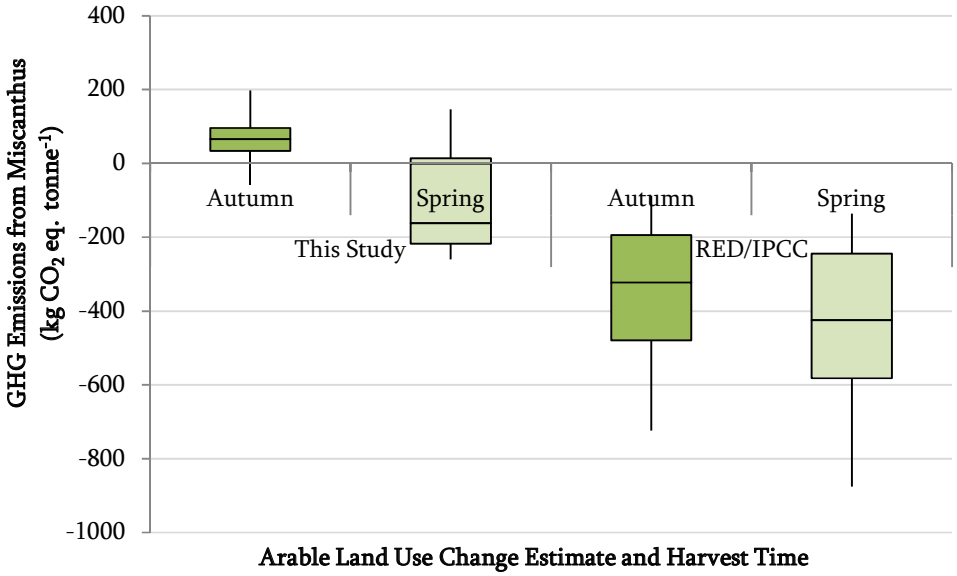

Figure 8-18. GHG emissions from direct LUC of arable land to Miscanthus comparing estimates from this study with that from the RED/IPCC.

The RED and IPCC methodology was used for observing conversion of grassland and forestland to Miscanthus (**Figure 8-19**). This estimated a modest increase in SOC with conversion of grassland, and a loss of 1.9 tonnes CO_2 eq. ha^{-1} $year^{-1}$ when forest land is converted. The results show a negligible decrease in GHG emissions is seen when grassland is converted to Miscanthus. Alternatively, a large GHG penalty can be expected if forestland is displaced.

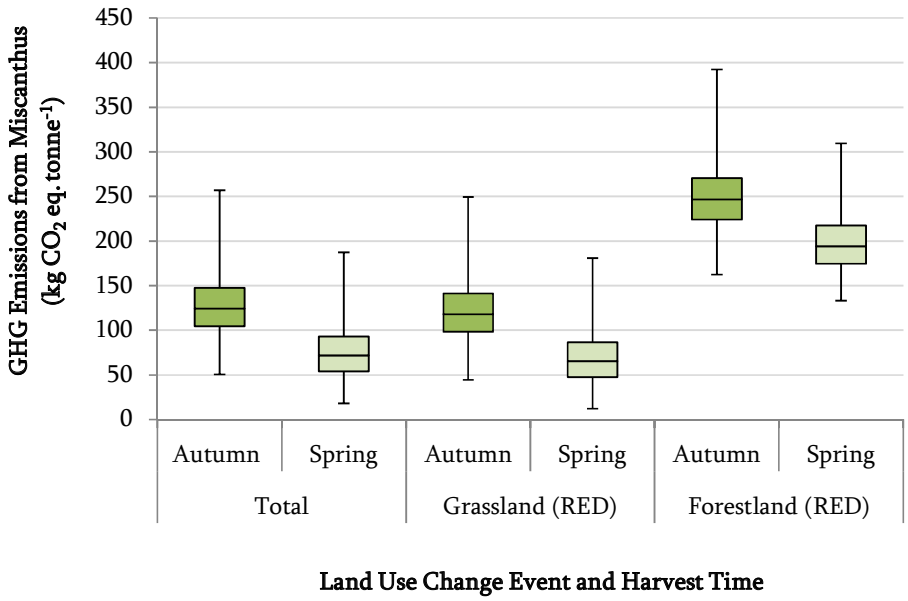

Figure 8-19. GHG emissions from direct LUC of grassland and forestland with Miscanthus.

8.4.3.3. Effect of Rotation Length on GHG Emissions

The effect of rotation length on GHG emissions from Miscanthus is examined here. The rotation length is assumed to be between five and 20 years. **Figure 8-20** shows how the overall GHG balance is affected. The results are based on the average application rates of fertilisers, use of fuel and yields.

The results show that the overall GHG emissions per tonne decrease by approximately 50% if the crop is maintained for 20 years in comparison with the shortest period of five years. Here the establishment and termination events are allocated between every tonne of Miscanthus that is produced on that site, therefore longer rotation length mean greater net yields, and thus establishment GHG emissions are reduced. Harvesting GHG emissions are generally constant throughout the year, although there is a two year lag between establishment and first harvest.

The sensitivity to rotation period shows that a decreased rotation time due to poor crop survival with early harvesting in autumn will increase the GHG emissions per tonne of Miscanthus. There is no data available on estimated rotation lengths with harvesting period. It must be noted here that the economic consequences of a reduced lifespan of the crop may be devastating as the establishment of Miscanthus is a major cost. It is vital to understand this effect if green harvesting is to be practiced.

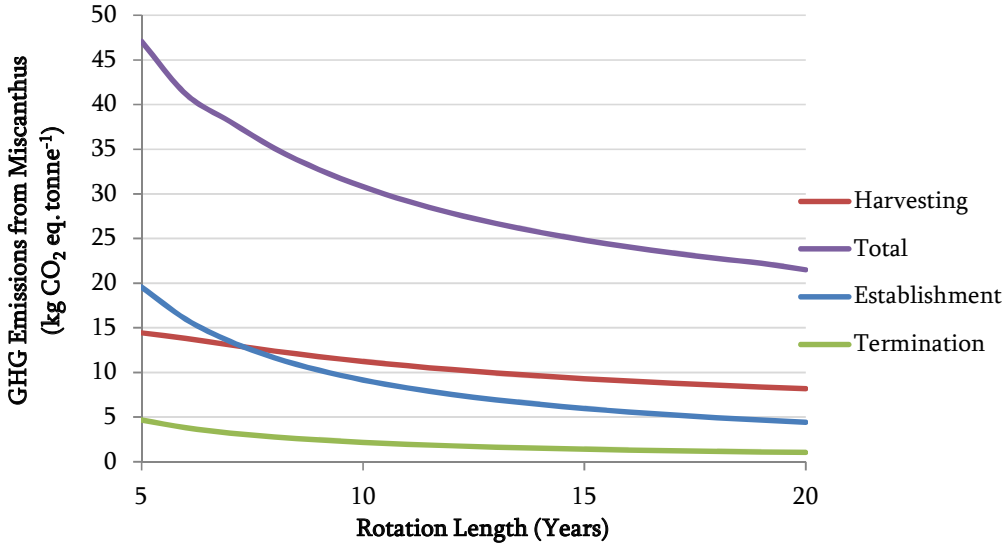

Figure 8-20. Effect of Miscanthus rotation length with GHG emissions per tonne.

8.4.3.4. Sensitivity to the Probability Density Function

In the Miscanthus cultivation case study, a uniform distribution was assumed for all parameters. As the results are apparently sensitive to the N fertiliser application rate, this assumption is tested here by re-calculating their application assuming a triangular distribution with a "best guess" of zero. The results are shown in **Figure 8-21** and demonstrate that the *pdf* has a small effect on the GHG emission results. A triangular distribution slightly reduces the variability of the final result compared to assuming a uniform distribution.

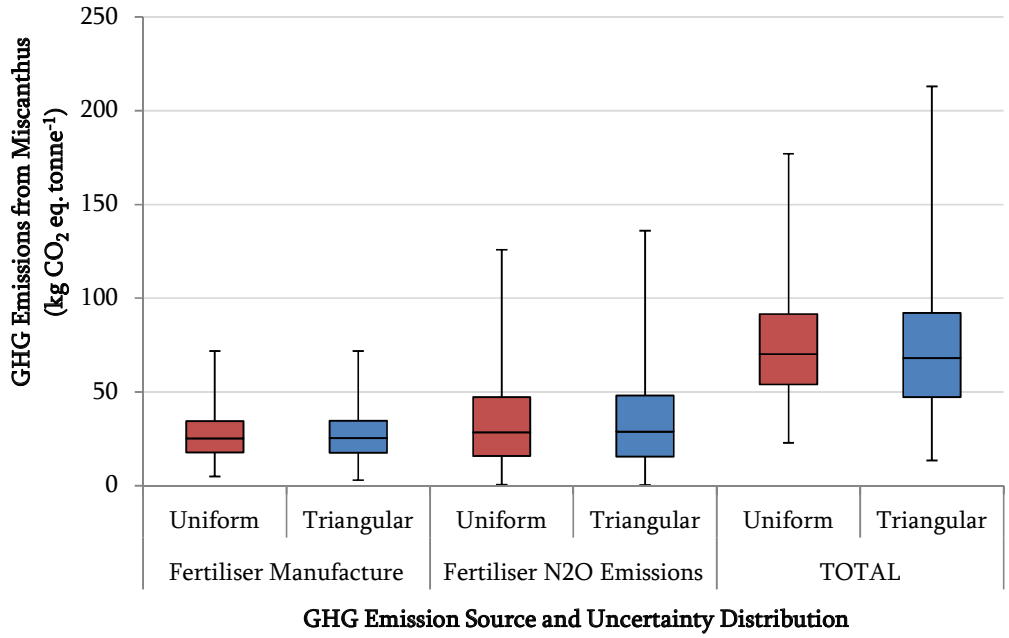

Figure 8-21. Effect of the pdf on the GHG emission results for spring-harvested Miscanthus.

8.4.4. Relative Impacts of Uncertainty and Variability

This section examines how these parameters vary due to variation or uncertainty, which is also performed for wheat cultivation in **Section 9.3.4** of this chapter. The results are also examined according to a series of runs; however these are modified compared to those used in wheat case study:

- Variation
    - Spatial variation (Run 1) – variation in yields
    - Temporal variation (Run 2) – variation in rotation length
- Uncertainty
    - Site inputs (Run 3) – fertilisers, pesticides etc.,
    - Manufacture of inputs (Run 4) – fertilisers
    - DLUC :Conversion of arable land (Run 5)
    - Fuel Consumption (Run 6)
    - N_2O emissions from soils (Run 7)

A further Run 8 will examine the combined effects of all uncertainty on the GHG emission results. **Figure 8-22** shows the results of these runs that result when Miscanthus is grown on arable land. Grassland and forestland conversion are considered to be unlikely cases and are not examined here, because the conversion of land has shown to causes large losses in SOC and are hence avoided. The results in **Figure 8-22** demonstrate the causes of variation in the results due to various sources of variability of inputs or uncertainty in emissions. A visual assessment of the array of runs indicates that Run numbers 3, 5 and 7 causes the majority of the variation in Run 8, which is the total variation in the GHG emission results. Runs number 1 and 4 also cause some variation in the results, whereas and Run numbers 2 and 6 are minor.

Run 3 shows uncertainty in the inputs to Miscanthus, which is mainly affected by assumptions on fertiliser use. Results from the sensitivity analysis showed that the overall GHG emissions are highly sensitive to these assumptions.

Run number 5 shows the uncertainty in the effects of DLUC on the GHG emission results. DLUC reduces the net GHG emissions from Miscanthus so that the majority of results are negative; indicating a net decrease in emissions due to an increase in carbon stocks on what was previously arable land. Again uncertainty could be reduced by using processed based models for estimating specific changes in SOC, such as Roth C (Hillier et al., 2009), or from more recent research into the carbon sequestration rates under Miscanthus (yet to be published). Run 7 again shows the large impact of N_2O emissions from soils.

Run 1 shows the variation in the results due to yield. The sensitivity of the results to yield is already shown in **Figure 8-17**. The annual yield of Miscanthus will affect the overall output from the site, so the results are somewhat sensitive to this. Run 4 shows how the GHG emissions from fertiliser manufacture affect the results. Overall, the impact of temporal variation is small (Run 2). This may be because the overall GHG emissions from the establishment and termination phase are relatively low.

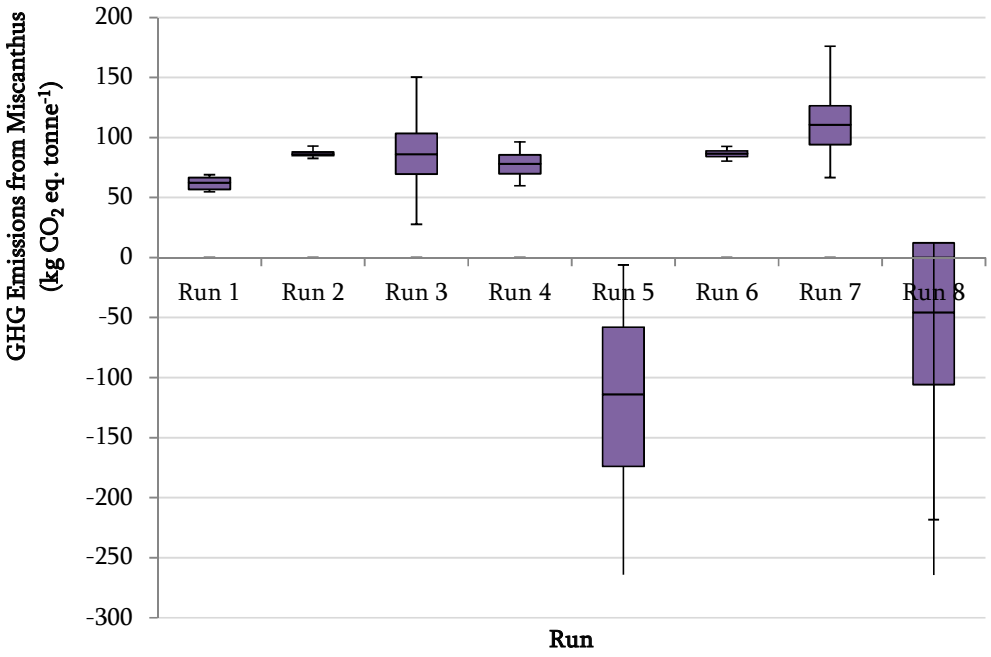

Figure 8-22. GHG emission results for Miscanthus cultivation in Runs 1-8. This includes DLUC due to conversion of arable land to Miscanthus.

8.4.5. Summary: GHG Emissions from Miscanthus

Miscanthus is a perennial energy crop that is harvested annually for biomass production. In the literature relatively high yields, low nutrient demands and ability to sequester carbon are expected to result in a low overall GHG balance for Miscanthus. There is evidence that the carbon sequestration rates under Miscanthus are lower than expected in the RED and/or IPCC as they do not consider the proportion of stored carbon that will remain after the crop is terminated.

Despite the rhizome multiplication phase is a laborious and intensive process, this research has shown that the process is 'GHG efficient', as the high yield of rhizomes means that the process has a negligible contribution on the total emissions of the crop.

There is an interesting anomaly seen in the GHG emission results regarding the rotation length of the crop. The GHG emissions from a tonne of Miscanthus halve when the crop is active for up to 20 years compared to ending the crop after five years, although the contractors stress that such a short duration would not make economic sense.

The results suggest that the GHG emissions from increased fertiliser application rates in autumn harvesting are not compensated by the higher yield achieved when harvesting in the autumn period.

The analyses of uncertainty, variability and sensitivity performed in this chapter on Miscanthus cultivation have identified the following major and minor sources of variability in the results:

- "Major" sources of variation
    - Yield
    - Fertiliser requirements and frequencies of applications
    - Fertiliser types applied
    - N_2O Emissions from soil
    - Carbon sequestration
    - Rotation lengths (at low rotations)
- "Minor" sources of variation
    - P_2O_5 and K_2O-based fertilisers
    - Diesel
    - Rhizomes
    - Crop residue deposition

8.5. Summary: GHG Emissions from Crop Cultivation

This chapter has shown that the GHG emissions from wheat and Miscanthus are highly variable due to a combination of variation in inputs and emissions due to:

- N application to soil
- N fertiliser manufacture
- Yield
- DLUC

In the literature, the application rate of fertilisers is highly uncertain in Miscanthus, though the commercial-scale growers claimed to never apply fertilisers to their sites, except one small application of potassium after five years growth. The expected GHG emissions from Miscanthus will be overestimated if literature resources are used.

The yield of biomass from a site has a major impact on the GHG emissions per tonne biomass. In wheat, there is an issue of allocation between wheat grain and straw. Wheat is particularly sensitive to increases in yields or value. As the yield of straw is lower (average 3.5 tonnes compared to 8.6 tonnes in wheat) and the price is much lower (£65 tonne compared to £198 $tonne^{-1}$), the sensitivity is lower in wheat straw.

Yields of Miscanthus also have a large impact on the GHG emissions per tonne; however the increased nutrient requirements for autumn harvesting are not compensated by the increase in yield. There is still some uncertainty in the long-term sustainability of harvesting Miscanthus in autumn, as this is purely a theoretical scenario.

The following chapter examines how some current GHG calculation tools calculate the GHG emissions from wheat. This facilitates the analysis of the effects of system boundaries and emission factors on the GHG emission results.

Chapter 9.

Assessment of GHG Calculation Tools

In light of concerns over climate change and the need for national inventories for greenhouse gas reporting, there has been a recent increase in interest in the 'carbon foot printing' of products. A number of LCA-based carbon reporting tools have been developed in both the agricultural and renewable energy sectors, both of which claim to accurately account for GHG emissions from arable cropping. These tools can make quite complicated LCA calculations assessable to those with less expertise in GHG reporting (**Figure 9-1**) and can help harmonise calculations to enable more reliable comparisons between products (Hennecke et al., 2012).

This chapter examines the results generated by these tools when providing consistent input data. This is novel work and has been published in:

A Comparison of Carbon Accounting Tools for Bioenergy and for Whole Farms. Whittaker, C., McManus, M. & Smith, P. *Environmental Software and Modelling,* 2013: 46 pp. 228-239.

This is presented in **Appendix 2**.

This analysis is performed in order to assess how variation in LCA results can still occur when variability in input data and of LCA methodology are fixed (**Figure 9-2**). The aim of this assessment is to examine how the system boundaries of tools may vary, and how they affect the final GHG emission results. The study also highlights how inconsistent emission factors can also have an impact on the GHG emission result.

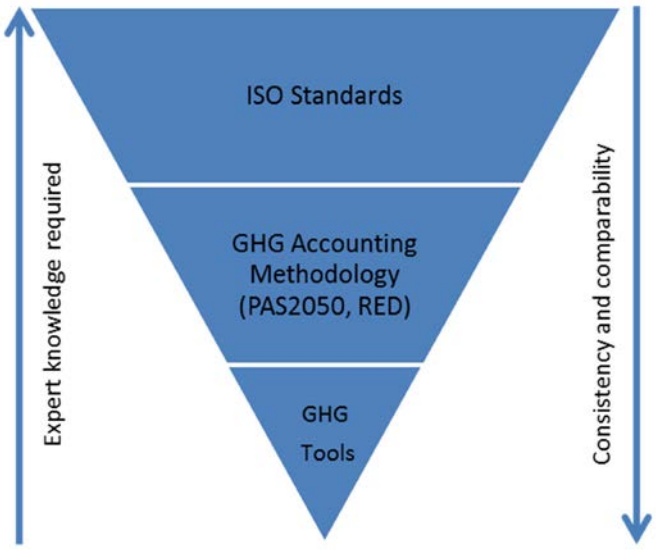

Figure 9-1. Hierarchy of knowledge requirement and relevance of LCA-based GHG tools for product assessment.

In this chapter a **'GHG accounting tool'** is defined as automated web, excel, or other software-based calculation tool that is developed for quantifying GHG emissions for production of a product (Colomb et al., 2012). The process of calculating emissions will depend on a series of underlying principles, methodologies, assumptions, and data that are built into the calculation mechanism of the tool (Hall et al. 2010). These are reviewed in this chapter.

First an introduction to the GHG calculation tools identified for study is provided. Once identified, a review of the goal and scopes of the tools is performed and the data requirements assessed. A sample dataset used in the case cultivation study are then adjusted to fulfil the demands of the least and highest 'data demanding tools'. The GHG emissions results are assessed and the causes of differences are determined. A 'multi-criteria analysis' (MCA) is performed to facilitate this assessment. A MCA will test how the tool's underlying goal and scope may affect the structure and accuracy of the tools, and hence their final GHG emission results.

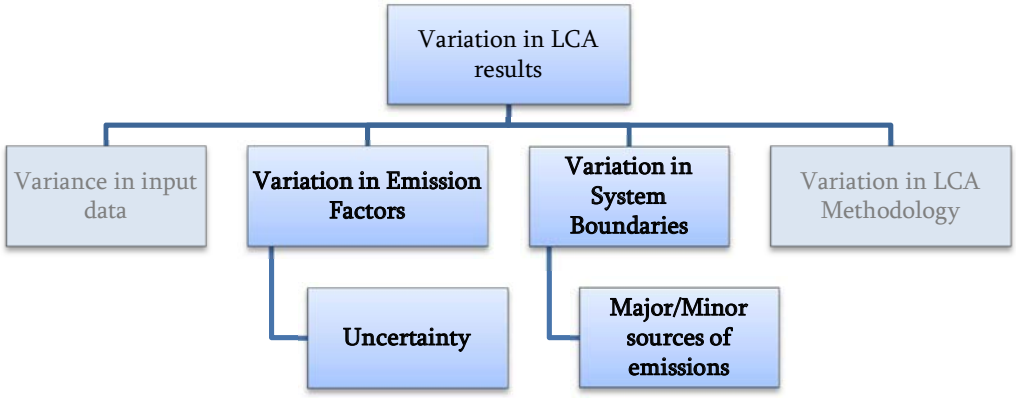

Figure 9-2. Focus of Chapter 10.

9.1. Identification of GHG Calculation Tools

A desk-based review was performed to identify existing GHG calculation tools, available in the public domain. The review focusses on tools applicable to the UK only, as the geographical range may affect the specificity of the tool and may be populated by country-specific emission factors (Colomb et al., 2012). Tools specific to calculating emissions from crop cultivation in the agricultural or bioenergy sectors were included in the study.

The desk-based review identified 31 resources that can be used to calculate the emissions from crop cultivation, or from a specific aspect of crop production. Sources originated from commissioned work to environmental consultancy groups, from governmentally and non-governmentally funded organisations, programmes developed from scientific research in Universities and from certification schemes.

Of the 31 resources, 15 were excluded as they are based in the United States (6), Australia (5), Europe (2), New Zealand (1) and Canada (1). Seven process-based models were excluded as they are typically used to examine specific processes in the soil. CPlan has two developed two tools, however the more recent version (CPLANv2) is not free to use. The Carbon Trust Carbon Footprint Expert Tool® (Carbon Trust, 2012) was also excluded from further review as this is only available to consultancies for a fee.

Eleven of these resources were selected for further review (**Table 9-1**).

Table 9-1. GHG Calculation tools selected for study.

Producer	Tool	Access	Final unit of measurement
Farm-Based Tools			
Country Land and Business Association	CALM	Online	Whole farm
SEE 360 Ltd	C-PLANv0	Online	Whole farm
Manchester University	CCalc	Download spread sheet	User defined
Climate Friendly Food	Organic Farmer Carbon Calculator	Online	Whole farm
Cool Farm Institute	Cool Farm Tool v1.1	Download spread sheet	Whole farm/ 1 hectare of farm/ 1 tonne of crop
Muntons	Muntons barley calculator v4	Download spread sheet	1 tonne of crop
Bioenergy-Based Tools			
Biograce	Biograce calculator v4b	Download spread sheet	hectare of crop/MJ bioethanol
Renewable Fuel Agency	RFA - RTFO Carbon Calculator v1.0	Download programme	hectare of crop/tonne/MJ bioethanol
North Energy Associates and AEA Technology	Biomass Environmental Assessment Tool	Download programme	hectare of crop/1000 litres bioethanol
Round Table of Sustainable Biofuels	RSB Tool	Online	hectare of crop/MJ bioethanol
Home Grown Cereals Authority	HGCA Biofuel GHG Calculator	Online	tonne/ litre/ GJ bioethanol

9.2. Main Accounting Tools Identified

In the 11 tools selected, six are designed in order to calculate the emissions on a farm-level and five examine cereal cultivation as part of the bioethanol production pathway, therefore they represent a combination of tools designed to raise awareness, and to report GHG emissions on a farm and product level. All of the tools require data input that would all be known by any farm owner or manager. These are reviewed below as it is important to determine the purpose of the tool before it can be assessed.

9.3. Farm-Based Accounting Tools

The following sub-sections provide a description of the GHG calculation tools that are specifically designed for assessing the GHG emissions of farms or crops. These are generally used to educate farmers about the main sources of emissions, evaluate mitigation projects, or report emissions to a consumer or certification board (Colomb et al., 2012). These tools adopt a streamlined LCA approach, examining GHG emissions to the farm or factory gate, or final product.

9.3.1. C-Plan Carbon Calculator

This is a web-based GHG calculator developed in 2007 by an agricultural consultancy in Scotland, whose goal is to provide an IPCC compliant calculator applicable to the agricultural sector. It aims to calculate, with as high a degree of accuracy as possible, the GHG emissions from a farm. The information will help make farmers aware of where emissions occur and are in a position to accept future GHG emission reduction challenges (C-Plan, 2007).

The authors claim that the tool follows Tier 1 IPCC calculations, however this is not clear for N fertiliser applications, as the data inventory sheet does not decipher between N and non N-based fertilisers, nor does it ask for the N content of the fertiliser. Also, there are no emissions attributed to organic fertiliser. Overall the tool is user friendly and provides an account of other sources of emissions not included here such as woodland and land use change. Transparency is lacking in the tool.

9.3.2. Climate Friendly Food Carbon Calculator

The goal of this tool is to accurately calculate the total GHG and carbon sequestration from farming activities on a whole farm (CFF, 2009). The target users are farmers who wish to account for their GHG emissions for marketing, economic or ethical reasons or to understand how they can adopt practices that lead to better quality soils. The tool is highly transparent, with references provided for each emission source.

Interesting, virtually no data is requested on N fertiliser use. Also, it appears that the methods in which emissions from farm machinery and implements are calculated are overestimated. By default, they attribute the whole construction of the machine to the single crop. This is a consequence of the tool accounting for the farm-level emissions over an unspecified period of time, rather than the emissions for one year's work. Here the emissions for farm machinery construction were readjusted so to not penalise the tool for this mistake.

The guidelines for the CFF Tool state that after ten years an item has 'paid off its carbon debt' meaning that the emission from manufacture do not need to be included, and farmers should use older machinery when possible (CFF, 2009). The logic is somewhat misguided, as although maximising the use of life of an item will reduce the relative emissions for 1 hour's work, older machinery is less likely to be fuel efficient and conform to current emission standards.

Overall the tool may over estimate emissions from machinery production, and also as it excludes N fertiliser application, this tool lacks comprehensiveness, therefore should not be used for assessing the GHG balance of arable crops.

9.3.3. Carbon Accounting for Land Managers (CALM) Calculator

This is a free online tool whose goal is to calculate the net balance of GHG emissions between farm activities and carbon sequestration due to management (CLA, 2008). Emissions are reported for each of the site inputs, including detailed breakdown of direct and indirect emissions from N application to soil. Overall, the tool is user friendly and provides a comprehensive account of GHG emissions from a farm although the transparency of emission factors is somewhat lacking.

9.3.4. Cool Farm Tool

The Cool Farm Tool is a farm-based GHG accounting tool which is designed with a farmer-focus but yet a high level of regional specificity (Pepsico, 2012). The goal is to accurately measure on-farm GHG emissions, so that the user can identify options in which to reduce overall GHG emissions.

There is a high degree of flexibility in the units and how data can be provided. Calculations for N_2O emissions from soil are based on Tier 3 IPCC approaches, which are based on published data for both soil-type specific and fertiliser-specific emission factors (Bouwman, 1996; Bouwman et al., 2002). The tool includes emission sources for all site inputs, including N from synthetic and organic fertilisers, and crop residues. Also, land use change and sequestration in soil and trees are included in the analysis.

The final emissions are reported in various ways, with a breakdown of emissions that occur, on both per hectare and per tonne bases, with graphical demonstrations also. Emission factors are visible and references provided, hence the tool is transparent. Also calculations are visible, though sometimes difficult to follow due to their complex nature, and the flexibility of the tool. Overall the tool is both comprehensive and transparent.

9.3.5. Muntons Farming Carbon Footprint Calculator

The Muntons Farming Carbon Footprint Calculator is designed for barley production; however the input data have been manipulated here to represent wheat. The tool was developed by the Low Carbon Futures and Stockholm Environment Institute and contains data from the Ecoinvent database (Muntons.com, 2012). The tool is farmer-focussed, therefore requires data input for standard site inputs. The tool is designed to calculate the emissions per tonne of grain, focussing on nine elements that are responsible for the majority of emissions from crop cultivation. This is done in order to help users identify the most effective methods in which to reduce emissions from farming (Muntons.com, 2012).

Data is required for the yield, hours of tractor use and main site inputs such as seed, fertiliser, organic fertiliser, and pesticide use. Emissions from urea manufacture are not included in the

data entry. The results are provided as a single figure per tonne of crop, without a breakdown of where the emissions occur.

9.4. Biofuel-Based Accounting Tools

The following sub-sections provide a description of the GHG calculation tools that are specifically designed for assessing the GHG emissions from biofuels. These also include the calculation steps necessary to examine the GHG emissions from crops. The tools are usually used to assess the GHG emission savings of biofuels compared to conventional fossil fuels.

9.4.1. HGCA Biofuels Greenhouse Gas Calculator

This is an online tool that is designed to provide a platform for calculation of credible GHG emission estimates from bioethanol production in the UK (Woods et al., 2005). The tool provides a set of default numbers that are based on expert knowledge of bioethanol production pathways, which can be adjusted by a user if required. Basic farm knowledge is required to calculate the emissions from a particular farm, and requires data on site inputs such as fertilisers, pesticides and also records the yield of grain and the yield and fate of straw. The results do not focus on a farm-level result, but are provided on either a tonne, litre, GJ of bioethanol, or petrol equivalent of bioethanol. Emissions for 1 hectare can be estimated by multiplying the emissions per litre for cultivation, by the yield of ethanol per tonne of grain (default is 374 litres $tonne^{-1}$) and the yield of grain per hectare. The supporting material clearly displays the emission factors assumed, with references; therefore the tool is generally transparent.

9.4.2. The BioGrace Project

The BioGrace Project was established after the publication of the RED (EC, 2009b), in order to provide a platform for harmonising biofuel GHG calculations to facilitate the implementation of the RED accounting methodology (Biograce.net, 2012). The project has developed an Excel spread sheet tool that contains emission factors and default calculations for biofuel production from 22 major feedstocks, including wheat, from cultivation to the final distribution of bioethanol. The default data and emission factors are not clearly referenced. Specific and highly detailed calculations for land use change, carbon stock changes and N_2O emissions are provided on separate pages, which are based on IPCC methodology and referenced. Overall the tool is moderately transparent in terms of methods of calculation of emissions, and comprehensive in terms of emission sources examined.

9.4.3. Department for Transport Carbon Calculator

The 'RTFO Calculator' produced by E4Tec for the Department of Transport, is designed to be used by biofuel producers to produce obligatory carbon and sustainability reports in order to be awarded with RTFO credits for GHG mitigation from biofuels. The programme is downloaded and installed and there are various options available to the user: they can develop fuel supply chains from scratch (which is not particularly user-friendly), or use default chains already populated with input data. The default data is based on data provided in Biograce. The emission factors can be modified by the user, and there is a lot of flexibility in terms of inputs and units.

9.4.4. Biomass Environmental Assessment Tool

The Biomass Environmental Assessment Tool (BEAT) was produced by AEA Technology and North Energy Associates on behalf of DEFRA and the Environment Agency in 2008, and was updated in 2010. The tool is designed to compare the GHG emissions from biomass supply chains for heat, power and biofuel production with fossil fuel alternatives and highlight the main sources of emissions to aid identification of potential mitigation options (AEA Technology and North Energy Associates, 2010).

The programme is populated with default data for various biomass supply chains, and it is possible for the user to modify some key parameters, particularly those that are responsible for a large proportion of emissions. The main calculations and emission factors are all referenced, and hence the tool is very transparent.

9.4.5. Roundtable on Sustainable Biofuels (RSB) Tool

The RSB Tool is a freely available online tool that enables users to calculate the GHG emissions of biofuel production (RSB, 2012). The tool can calculate emissions based on the methodology developed by the RSB (RSB, 2011), or the RED methodology (EC, 2009b). Following the RSB methodology, emission factor data is provided by Ecoinvent for site inputs, and emissions of N_2O emissions from fertiliser application are based on the IPCC (De Klein et al., 2006). Calculations include GHG emissions arising from infrastructure development and delivery, whereas in the RED methodology-based tool these are excluded.

The tool can be used to calculate the GHG emissions from a single stage of the production of biofuels, and contains a detailed section on site inputs to cultivation. Virtually no default data is provided in the tool, therefore the user must provide detailed information on the site, including the average rainfall and soil taxonomy. The detailed manual and 'help points' throughout the tool aid completion of the required data. If the full data cannot be provided the tool calculates emissions based on a lower IPCC tier. The tool also includes an option to calculate GHG emissions from direct land use change and changes in management.

9.5. Summary of Tools

The main aim of the majority of the farm-based tools is to educate farmers as to where emissions occur on their farm (Muntons.com, 2012) and identify GHG mitigation options so that they are in a position to accept future GHG emission reduction challenges (C-Plan, 2007). This can be for marketing, economic or ethical reasons or to understand how they can adopt practices that lead to better quality soils (CFF, 2009). The goal of the bioenergy-based tools focuses on accurately measuring the GHG balance of various stages of the biofuel supply chain.

Data requirements vary considerably across the tools. Some are populated with default data that the user can review and change if necessary, whereas some are completely dependent on the user having sufficient data to complete it themselves. The following section discusses the system boundaries of the tools, which will affect the amount of data that must be supplied.

9.6. System Boundaries of the Tools

The tools vary considerably in their system boundaries, or their 'scope' of the GHG assessment (**Table 9-2**). As there are differences in the system boundaries in the tools, there will be differences in the amount of data that must be collected in order to use the tool effectively.

Diesel is the only emission source consistent in all tools. From the analyses performed in **Chapter 9**, it was concluded that the nitrogen fertiliser, N_2O emissions and direct land use change (DLUC) were the most significant sources of GHG emissions from one hectare of wheat, so tools should at least account for these parameters.

Table 9-2. Summary of system boundaries of the GHG calculation tools.

	Farm-Based Tools							Bioenergy-Based Tools					
Tool	C-Plan	CCaLC	CFF	CALM	Cool	Muntons	HGCA	Biograce	BEAT	RTFO	RSB	RSB -RED	This Study
Wheat Yield	✔		✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
Diesel	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
N_2O (Fertiliser)				✔	✔		✔	✔	✔	✔	✔	✔	✔
Fert. Manufact.	✔	✔		✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
N_2O (Residues)	✔		✔	✔	✔			✔	✔		✔	✔	✔
Manure N_2O				✔	✔			✔			✔	✔	✔
Seeds		✔				✔	✔	✔	✔		✔	✔	✔
Machinery			✔						✔				✔
Pesticides		✔			✔	✔	✔	✔	✔	✔	✔	✔	✔
Electricity	✔	✔	✔	✔	✔		✔		✔	✔			✔
Manure[a]		✔	✔						✔		✔		✔
DLUC		✔	✔	✔	✔			✔		✔	✔	✔	✔
Straw Yield			✔		✔		✔		✔		✔	✔	✔
Sequestration[b]	✔		✔	✔	✔								
Buildings			✔								✔		

a. Delivery of manure

b. Sequestration in the soil and in farm reserves (e.g. woodland)

9.7. Multi-criteria-analysis of relevant accounting tools

A multi-criteria-analysis (MCA) was performed, following the methodology described in Hall et al. (2010), to test for relative weaknesses and strengths in the tools identified. This will also help identify whether differences in the results are due to system boundaries or inbuilt emission factors.

The MCA adapts the criteria used in Hall et al. (2010). These objectively assess each tool whether they were:

- User friendly
- Informative
- Transparent
- Comprehensive

An example of the MCA criteria and scoring process is shown in **Table 9-3**. Comprehensiveness was based on whether or not the tool includes the farm inputs that most sensitively affect the GHG emissions from a crop. The analysis did not necessarily assess the accuracy of the results, as this may be open to debate without the appropriate experimental data available to determine accuracy.

Table 9-3. Example of criteria questions and scoring in the MCA.

Criteria/Questions	Score			
User Friendliness	3	2	1	0
Is the tool readily available?	Yes- Online or ready to download	Yes - but requires installation	Yes- but requires permission or password	Not available
Is support available?	Yes- A support telephone number	Yes - A support email address	Less obvious	None
Are manuals provided?	Yes - Detailed with data collection guidance	Yes - detailed manual	Basic instructions	None

Using IPCC Tier 1 emission factors was regarded as a suitable baseline to account for N_2O emissions from fertiliser, crop residue incorporation and manure application, and tools applying this were given a comprehensive score of 2. Those adopting a higher Tier approach (2 or 3) were rated higher. Following the approach by Hall et al. (2010), the categories were kept separate as not to imply they are equally weighted in importance in performing a GHG assessment of wheat. The criteria were assessed according to four levels of relevance (an example given in **Table 9-3**), and each tool was given a total score for each category.

9.8. Application of calculation tools to a case study

This study aims to examine whether there is variation in the GHG emission results calculated by the tools. It is hypothesised, that if the input data is fixed, and LCA methodology is excluded, then any differences in the results are due to the systems boundaries, or the emissions factors assumed within the tool. A random sample farm was used for input into the tools. To satisfy the data requirements of all tools, some additional data is required on climate and soil types, and the data was collected from literature. The dataset used is presented in **Table 9-4**.

Two tools require more detailed on the specific locations of the model farm. The Cool Farm Tool requires broad climatic data to be listed, as well as some basic soil parameters such as texture, SOC content and pH. The RSB tool requires more detailed soil type data as well as the input of meteorological data. Both these tools use the data to generate more specific IPCC Tier 3 emission factors for N_2O.

To eliminate methodological variation, the final emissions were based on a 'per hectare' basis, assuming that all straw is incorporated into the soil. Any differences in how emissions are allocated between wheat and grain will not be observed.

Table 9-4. Sample data used for analysis of GHG tools.

Input	Input/Parameter	Amount	Units per ha	Reference
Site Establishment	Diesel	71.8	litres	Farm sample
Seed	Wheat grain	175	kg	
Fertiliser	Urea (46.4% N)	123	kg	
	Ammonium Nitrate (34.5% N)	330	kg	
	Cattle Slurry(3% N)	633	kg	
	(Total N)	190	kg N	
	Phosphate	60	kg P	
	Potassium	43	kg K	
Pesticides	Total Pesticides	3.81	kg a.i	
Harvesting	Diesel	20	litres	
Machinery Uses	Hours	3.3	hour	
Soil Data	Soil Texture	Medium		Select middle-range types
	Soil Organic Matter	1.72<=5.16	%	
	Soil Moisture	Moist		
	Soil Drainage	Good		
	Soil pH	5.5 <= 7.3	pH	
	Soil Type	Inceptisol		(USDA, 2005)
Average rainfall		840	mm	1971-2000 average for England (Met Office, 2012b)
Yield Data				
Wheat grain (@13.5% m.c)	7.7		tonnes	Farm Sample
Straw	4		tonnes	

9.9. Results: Multi-Criteria-Analysis

The MCA showed differences between the tools for each of the four categories assessed. Across the four categories, the results differed between farm and bioenergy-based tools (**Table 9-5**). **These differences may be due to the goal of the tools.**

The main goal of some of the farm-based tools is to provide a calculation platform to educate farmers about GHG emissions occurring due to their activities and choices of management. In bioenergy-based tools the goal and scope is to accurately assess the GHG savings compared to using conventional fossil fuels.

Farm tools achieved a higher score for 'user friendliness' and the bioenergy tools were generally rated more informative, comprehensive and transparent. The results of the MCA in some way reflect the main goal and scope of the tools. As it is more likely that farm-based tools will be used by non-LCA practitioners there is a level of user-friendliness expected (Colomb et al., 2012).

Bioenergy-based tools are generally used by LCA practitioners, or those with expertise in the industry to assess whether a biofuel has reached its GHG saving target. These demand a greater level of information, transparency and accuracy. In terms of user friendliness, the highest rating tools were the CFF and CCalc Tools (78%), followed by the Cool Farm Tool and the HGCA tool (72%). These tools were rated accordingly due to ease of access, intuitiveness, flexibility of input units and support and guidance for using the tools. Poorer performing tools were lacking instructions and flexible or non SI units.

In the 'informative' category the majority of tools were high scoring. The highest rating tool was the Cool Farm Tool (100%), mainly because it provides results in various formats and with a breakdown of all emission sources. Less informative tools did not provide a clear enough breakdown of emissions.

The highest rated tools in terms of transparency were the Cool Farm, BEAT and RSB Tools. The Cool Farm and BEAT Tools permit the user to access the original Excel-based calculations, including referenced sources of emission factors. The RSB Tool does not provide an Excel-based model, although the manual is highly transparent. Lower ranking tools did not provide sufficient details of the types of emission sources included in the analysis, or provide details on the sources of emission factor data.

The most comprehensive tools were the Cool Farm and RSB Tools, as these included LUC and adopted Tier 3 IPCC methodology. LUC was included in six out of 11 tools, and these were given a higher rating as LUC can potentially dominate GHG emissions in agricultural LCA's (Roches et al., 2010). The least comprehensive tools did not state whether N fertiliser manufacture or application were included. Only BEAT included some aspects of uncertainty.

Table 9-5. Results of the MCA for the GHG calculation tools assessed (percentage scores).

Category	Farm-Based Tools							Biofuel-Based Tools					
	C-Plan	CFF	Muntons	Cool Farm	CALM	CCalc	**Average**	RSB	Biograce	HGCA	BEAT	RTFO	**Average**
User Friendliness	67	78	56	72	67	78	**69**	61	56	72	39	61	**58**
Informative	50	83	17	100	83	50	**64**	83	83	67	83	67	**77**
Transparency	17	50	17	83	75	50	**49**	83	83	42	83	42	**67**
Comprehensiveness	20	10	23	90	60	33	**39**	90	80	37	40	43	**58**

9.9.1. Results: Effects of the Systems Boundaries on the GHG Emission Results

Figure 9-3 shows the overall emission profile generated by each tool. This shows the result generated by each tool when inputting the data from **Table 9-4**. The emissions from LUC are examined separately as they are usually large and would distort the graph.

The results show considerable variation in the results across the tools due to different system boundaries. For example the CFF tool has not included emissions from nitrogen fertiliser manufacture and N_2O emissions from soil, which account for over 60% of emissions in the tools that include them. The tool has a much lower emission profile than other tools. Here it can be seen that the difference in the system boundaries of the tools can affect the results by a large amount. The tool will not provide an accurate account of GHG emissions from arable farms and therefore should be avoided.

There is also variation between tools when the system boundaries are consistent. For example all of the biofuel-based tools include nitrogen fertiliser manufacture in their system boundaries, but each has calculated a slightly different GHG emission rate. In the C-Plan and CALM tools the different is quite large. It can be seen in the results that the difference in emission factors used in the tools can also affect the results by a large amount.

The results from the tools are also compared with the results of this study. It appears that the results are comparable with those calculated by bioethanol tools, but only when the default of 1% N_2O-N kg N^{-1}. Including a higher range in the uncertainty analysis increases the overall estimated GHG emissions. This is discussed in more detail in the following sub-sections.

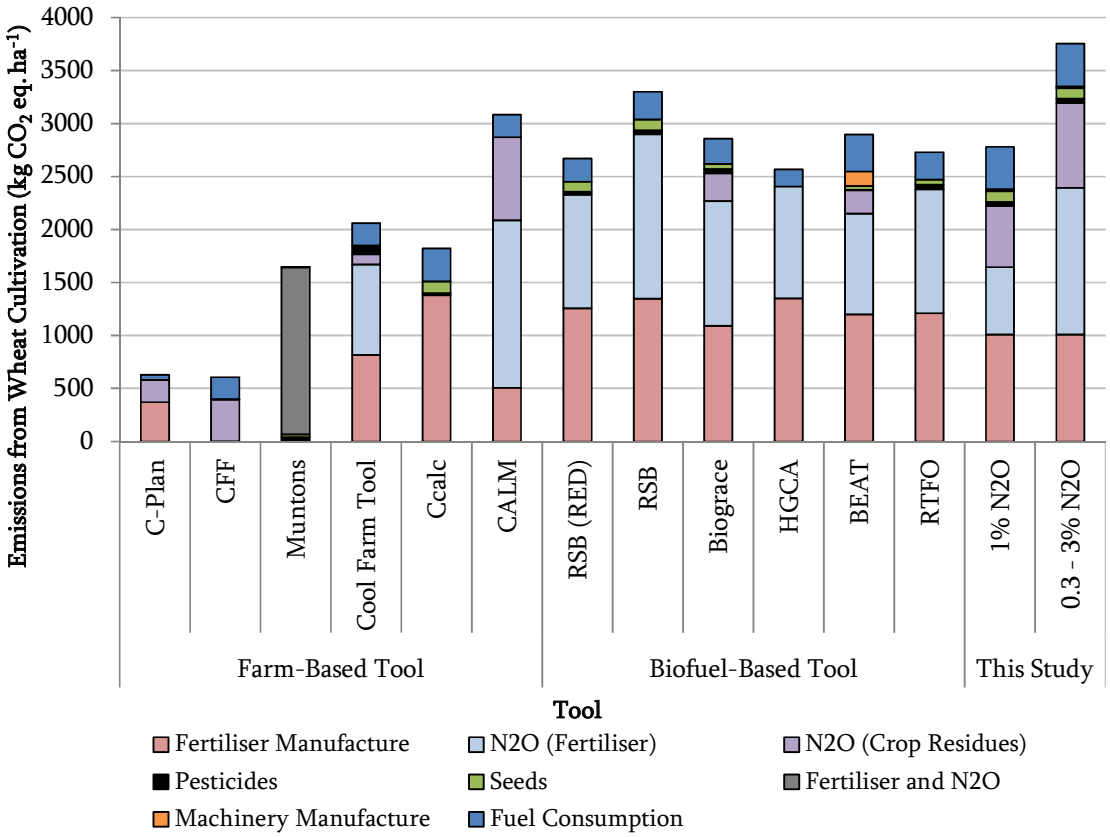

Figure 9-3. Emission profile from each GHG accounting tool for 1 hectare of wheat cultivation.

9.9.2. Farm vs. Bioethanol Calculators

Overall, on a per hectare basis, the total GHG emissions ranged between 606 and 3298 kg CO_2 eq. The average emission result calculated by all tools is 2239 kg CO_2 eq. ha^{-1}, where the average for the farm tools (1642 kg CO_2 eq. ha^{-1}) is almost half that of the bioenergy tools (2836 kg CO_2 eq. ha^{-1}). This is mainly due to the lower results calculated by the C-Plan and CFF Tools.

The farm-based tools scored a relatively low score in comprehensiveness, suggesting that their goal to 'accurately measure the GHG emissions from a farm' is not achieved in some of them. This is particularly true for the CFF Tool, which achieved the lowest score (10%) as it excludes nitrogen fertiliser manufacture and N_2O emissions from soil. The C-Plan Tool was rated low (20%) due to both a lack of detail on which GHG emission sources are included. They do not state which sources of N_2O they account for and their estimate for fertiliser production is lower than expected. The tool has a separate entry for 'crops' in the results; however it is not clear what this specifies. Both the CCalc and Muntons tools have also been penalised in the comprehensiveness score due to the level of transparency on N_2O emissions from soil.

The results of the bioenergy tools show greater consistency in both the total GHG emission result and the system boundaries. These tools also have a higher score in comprehensiveness than the farm tools. This may be due to way that the bioenergy tools are used. They are designed to facilitate the calculation of GHG savings from biofuels; therefore it could be argued that accuracy is of a higher importance than in the farm-based tools. Also there are some GHG reporting methodologies available which prescribe the sources of emissions that must be accounted for. These factors may contribute to why these tools appear to be more homogeneous in both system boundaries and the GHG emission results.

9.9.3. Sources of Variation in the Results

Figure 9-4 shows the causes of variation in the GHG emission results that can be deduced from the level of transparency in the tools. Some tools provide details of the calculations, whereas some could be described as 'black boxes' where only the inputs and outputs are visible (Carvalho et al., 2012).

Again, the results show the importance of DLUC, N_2O emissions from soil and nitrogen-based fertiliser. The results show that differences in the way these GHG emissions are calculated can cause large differences in the results. This was also evident in the results from **Chapter 9.** These two sources of GHG emissions are discussed in the following sub-sections.

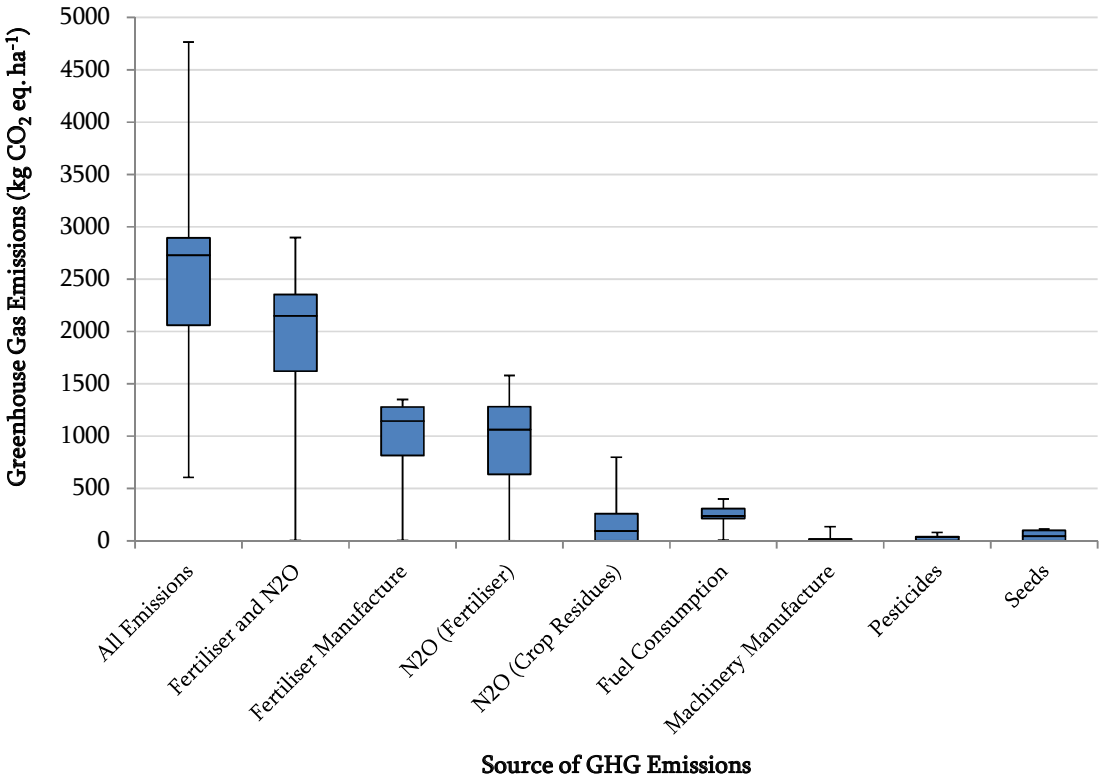

Figure 9-4. Sources of variation in the overall GHG emissions for wheat cultivation across GHG calculation tools.

9.9.4. Nitrogen Fertiliser Manufacture

Fertiliser manufacture is identified as a major source of variation in the tools. Often, there is a lack of transparency in the tools, making it difficult to determine which emission factors have been used. Where not stated, the emission factors can be deduced by manipulating the tools so that nothing but 1 kg nutrient was added to the site. The results from this are provided in **Table 9-6**.

It can be seen that some tools do not provide any evidence of the source of their emission factors. These tools generally scored low on the transparency category. Only 4 of the 11 tools provided separate emission factors for ammonium nitrate and urea, despite that emissions from ammonium nitrate are approximately 67% higher than urea (Brentrup & Palliere 2008). Estimates of P_2O_5 and K_2O are more consistent.

Table 9-6. Emission factor estimates for fertiliser manufacture across tools.

	Emission Factor per kg Nutrient (kg CO_2 eq. kg^{-1})					Reference Cited
Tool	'N Fertiliser'	Ammonium Nitrate (N)	Urea (N)	Phosphorous (P_2O_5)	Potassium (K_2O)	
C-Plan	0.63[ac]	-	-	-	-	None
CFF	-	-	-	-	-	None
Muntons	9.21[bc]	-	-	2.2[c]	0.5[c]	None
Cool Farm	-	6.20	1.48	1.3	1.5	EFMA 2000b; EFMA 2000a
CALM	-	3.80	1.24	-	-	None
CCalc	6.98	-	-	1.86	1.77	North Energy Associates 2006a
RSB		8.55	3.30	2.02	1.44	EcoInvent 2007
RSB (RED)		8.16	3.07	1.73	1.12	Biograce.net 2012
Biograce	5.88	-	-	1	0.6	EUCAR et al. 2006
HGCA	6.69	-	-	0.71	0.46	None
BEAT	6.92	-	-	1.85	1.76	North Energy Associates 2006a
RTFO	5.92	-	-	1.01	0.58	None
This study (Average)		8.55 3.00[c]	3.30	2.02	0.50	EcoInvent 2007

a. This is per kg of fertiliser.
b. Includes N_2O emissions from soil.
c. BAT production based on Ecoinvent data modified using industrial data.

9.9.5. Nitrous Oxide Emissions from Soil

Emissions of N_2O emissions from N fertiliser application, crop residue incorporation or manure application are identified as another major source of variation in the tools. As with fertiliser manufacture, there is also a lack of transparency in the tools, making it difficult to determine the sources of N_2O that are included in tool. The issue could be improved if details of the system boundaries of the tools were stated on the tool developers' websites or manuals. Alternatively, tools should provide disaggregated results for emissions from soils.

To examine the total N_2O emissions from each tool, they are compared with those expected according to the IPCC calculation methodology. The results are shown in **Figure 9-5**. It shows that there is large variation in the results. The C-Plan and CFF tools have appeared to greatly underestimate the N_2O emissions occurring on the model field, while the CALM tool appears to have over-estimated them. All the other tools deviate fall within a similar range.

Many of the tools state that they follow IPCC Tier 1 emission factors, and few have applied Tiers 2 or 3, though the details of the calculations are not transparent. It is suggested that a Tier 3 approaches are more appropriate for accurately assessing N_2O emissions from a particular site (Whitaker et al., 2010), and this is adopted by both the Cool Farm and RSB Tools. It is also understood that a Tier 3 approach can reduce uncertainty in the results (Guo & Murphy 2012), however way the tools present their results this detail would not be noticed.

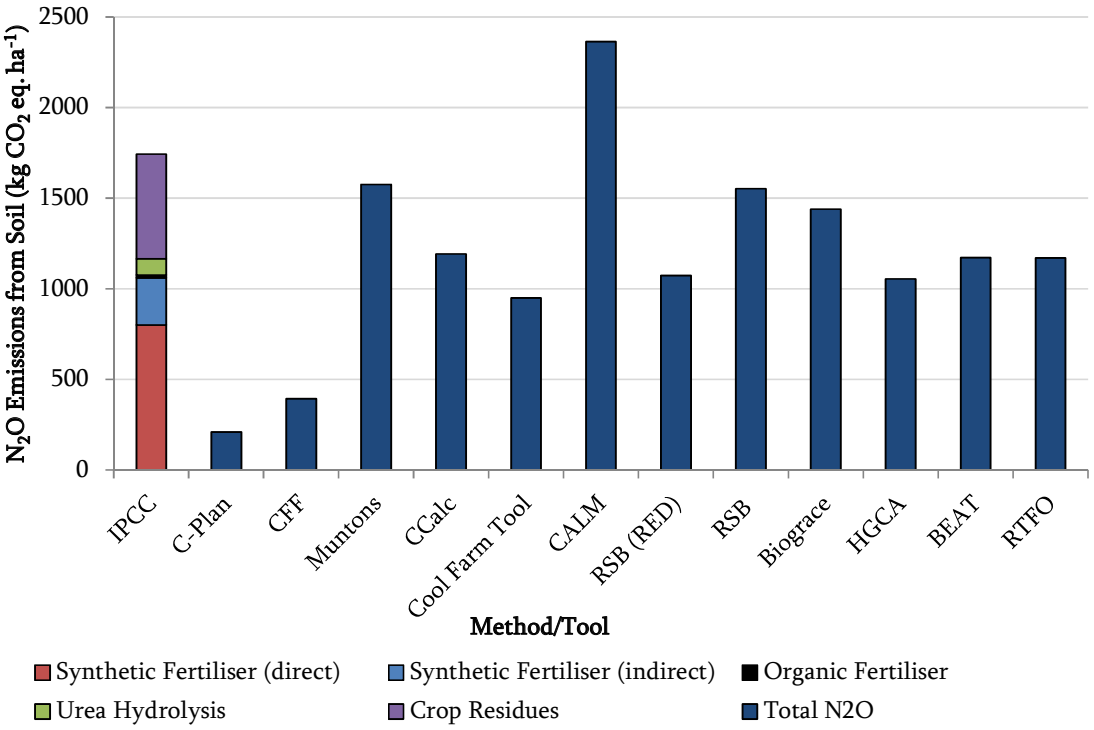

Figure 9-5. Expected and observed GHG emissions from soil due to fertiliser application in the sample study.

Three tools apply higher Tier methods for calculating N_2O emissions from soil, yet use different resources to do so. The Cool Farm Tool utilises emission factors derived from Bouwman (Hillier et al., 2012, 2011), the RSB uses Ecoinvent (Nemecek et al., 2007) and the IPCC (Tier 2) when following the RED methodology (De Klein et al., 2006). The results suggest that there is still variation in the calculated emissions. Some validation of the most appropriate IPCC Tier 3 emission factors may be required for purposes of harmonisation, and it is expected that these emission factors will depend on location.

There is also a lack of transparency with regards to N_2O emissions from crop residues also contribute to the overall GHG emissions from wheat cultivation. These were clearly included in 6 of the 11 tools and the magnitude ranged between 96 and 782 kg CO_2 eq. ha^{-1}, or between 5% and 33%. Some tools require the user to specify the fate of straw, i.e. how much was incorporated or removed, and some (e.g. BEAT) assumes that straw is removed from the site. Differences in this aspect of the calculation may be another cause of the variation of the results.

9.9.6. Direct Land Use Change

Direct land use change is another very significant source of GHG emissions from crops, which was also found in the analyses of the GHG emissions calculated for the case studies in **Chapter 9**. In the sample of tools, four did not include details on DLUC (C-Plan, Muntons, HGCA, and BEAT). Where it is included, there is still variation in the estimated result, though the range is greater for DLUC of forestland to arable land (**Figure 9-6**).

The order of magnitude of GHG emissions from wheat ranges between 606 and 3298 kg CO_2 eq. ha^{-1}, whereas LUC ranges between 1918 and 7000 kg CO_2 eq. ha^{-1} or 4147 and 27000 kg CO_2 eq. ha^{-1} for grassland or forestland conversion to arable land, respectively. The ranges of potential impacts of LUC are highly uncertain, but generally large. The tools estimate different GHG implications of LUC, despite the Cool Farm Tool, Biograce and RSB Tools using the same original resource to calculate emissions (Bickel et al., 2006).

For data entry of details of LUC, some tools require a selection of 'before' and 'after' land uses, whereas some require more details on the soil type, geographic zone and changes in how the site is managed. Out of the 7 tools that include LUC, the majority provide default drop-down menus as well as offering the user-defined inputs. Both the Cool Farm and RSB Tools provide a detailed calculation tool for LUC, including changes in tillage, inputs and residue management. The RSB Tool does this, although in a generally less user-friendly manner. Though it is outside the scope of this research to determine the accuracy of LUC estimates, it is clear that the Cool Farm Tool offers both the most comprehensive and most accessible calculations for LUC for a non-LCA practitioner to use, though if the user can calculate their own LUC estimate then all tools that offer this function are appropriate.

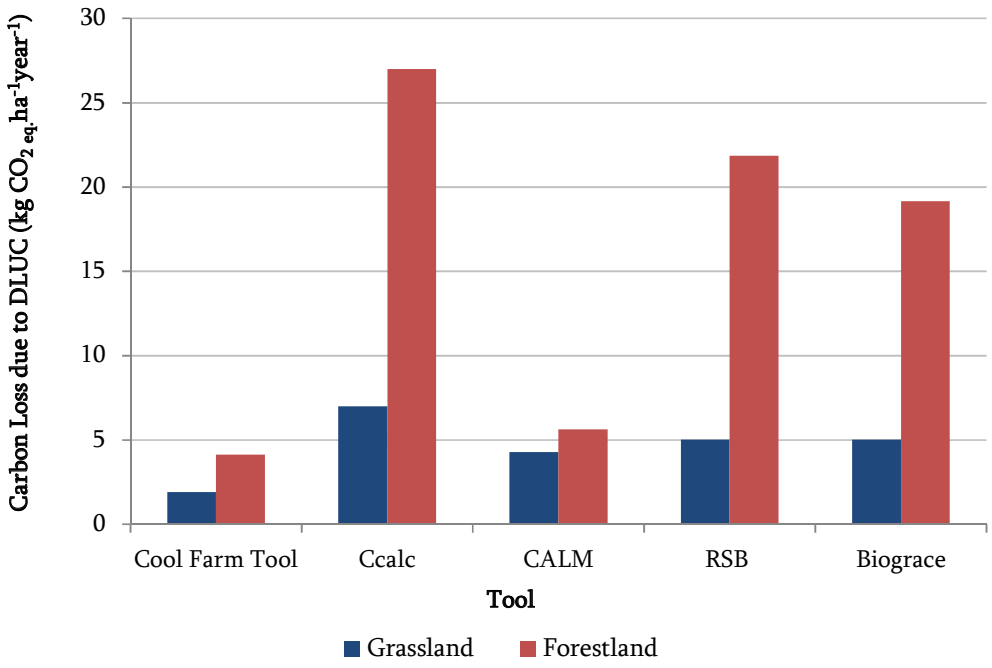

Figure 9-6. Variation in estimates for LUC from grassland and forestland to arable land.

9.9.7. Uncertainty Analyses in the Tools

As all but one tool provide a single result from one hectare of wheat, uncertainty was clearly lacking. Providing a single result, rather than a range or level of uncertainty, suggests a lack of comprehensiveness as it provides some information on the robustness of the data sources used and detail any temporal or spatial uncertainty (Guo & Murphy 2012). The result is limited in that it will only give users an indication of the average GHG emission for their crop or a baseline from which mitigation projects can be compared.

Only BEAT included some indication on how the result may range, however they do not provide details of how the uncertainty assessment was provided and the overall range is very small. The CCalc Tool requires the user to define the level of data quality, but it does not appear to be referred to in the results.

9.9.8. Emissions from Manure

In the sample of tools, four (C-Plan, CFF, CCalc and BEAT) included the emissions from manure delivery. There is evidence that the majority of other tools excluded such emissions, as they are not 'asked for' in the input sheet of the tool, or they are not reported in the emission results. These tools may have excluded manure from the calculations as they could be considered to be are a relatively minor source of emissions, compared to artificial fertilisers.

Some justification of the exclusion is that the N_2O emissions are attributed to the meat sector, in the waste disposal phases of animal husbandry. The tools that exclude emissions from

manure may assume that the emissions are accounted for in the meat sector. The IPCC calculation methodology specifies, however, that all manure and organic fertiliser application should be attributed to the crop that receives it for fertilisation purposes; therefore this should be included in all tools.

9.10. Summary

In summary, transparency varies across tools and there is evidence that the majority of the variation in GHG emissions is caused by fertiliser manufacture and N_2O emissions, which represent over 60% of total emissions in those tools that include them. There is evidence that the estimates for GHG emissions from fertiliser manufacture vary because the tools use different references (**Table 9-6**).

In terms of the actual GHG estimates, emissions from LUC are the largest sources of variation, but this was not featured in all tools. Emissions of N_2O from soil and fertiliser manufacture are also significant sources of variation between the results calculated. As transparency is lacking, it is difficult to determine whether this is a result of the system boundaries of the study or the calculation methodologies employed. The variation in fertiliser manufacture is due to a combination of data sources used by each tool, and as it is typically a major contributor to total GHG emissions from an arable crop, small differences can generate very different results.

Farm-based tools are more user-friendly, though bioenergy-based tools are more informative, transparent and comprehensiveness. An exception to this is the Cool Farm Tool, which was the highest rated in the MCA. The differences in scoring between the selected farm and bioenergy tools may be due to their original goal and scope: farm-based tools are generally designed to inform users on the sources and mitigation options on a farm level; bioenergy-based tools provide information on the GHG emissions from producing a single crop. The differing goal and scope of the two approaches may affect the design of the subsequent GHG calculation tool, and hence the results.

This study has demonstrated that different goal and scopes, system boundaries and underlying emission factor data within GHG calculation tools can result in very different results despite the same input data used. The goal and scope of a tool is the most important factor in determining its intended use. Bioenergy-based tools demonstrated less variation across the results than farm-based tools, which may be due to the methodological guidelines available for biofuel reporting. Therefore, there is a need to harmonise both methodology and emission factors in biofuel GHG calculation tools, so that at least a consistent result can be generated.

The following chapter examines the effects that decisions on LCA methodology have on the GHG emission results.

Chapter 10.

Assessment of LCA Methodology

In order to ensure that GHG emissions are actually reduced by biofuels, it is vital that the GHG balance of producing and delivering them is favourable (Black et al., 2011).The sustainability implications of biofuel supply chains can vary significantly, depending on the biomass feedstock, the production process, and by how the GHG emission balance is calculated (Royal Society, 2008). A number of GHG reporting methodologies have been developed in order to provide a framework for LCA calculations.

This chapter examines the effect of LCA methodology on the GHG emission results for 1st and 2nd bioethanol production. This is novel work and has been published in:

Greenhouse gas reporting for biofuels: A comparison between the RED, RTFO and PAS2050 methodologies. Whittaker, C., McManus, M. & Hammond, G. *Energy Policy*. 2011: 39 pp. 5950-5960.

This is paper presented in **Appendix 1**.

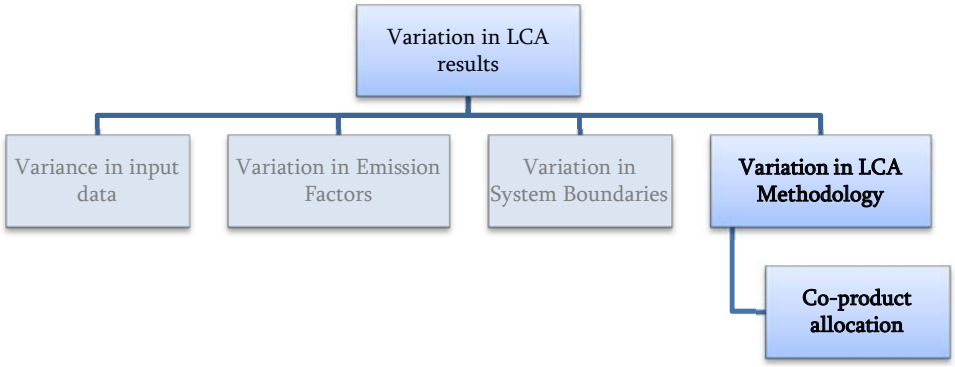

Figure 10-1. Focus of Chapter 11.

The aim of this chapter is to review the LCA methodological approaches adopted in current GHG reporting methodologies that can, or could be used for bioethanol production. The implications that different methodologies have on the final GHG emissions from bioethanol are examined (**Figure 10-1**) in the context of the GHG emission saving targets specified in the RED.

First this chapter reviews the goal and scope of the RED, RTFO and PAS2050 GHG reporting methodologies. A review of the LCA methodology applied in these will be performed. Finally the GHG emission results for 1st and 2nd generation bioethanol production will be assessed using the model datasets from the wheat grain, straw and Miscanthus case studies described in **Chapters 6, 7 and 8.**

10.1. Goal and Scope of the GHG Reporting Methodologies

Chapter 2, **Section 2.6** introduces three GHG reporting methodologies adopted in policy and legislation in the UK:

- The Renewable Transport Fuel Obligation (RTFO).
- The European Union (EU) Renewable Energy Directive (RED).
- The Publicly Available Specification 2008:2050 (PAS2050).

The RED and RTFO methods were developed in response to concerns over biofuel sustainability, and are set in place to regulate the GHG emissions that occur from the production of biofuels. The PAS2050 methodology was developed in response to a "broad community and industry desire for a consistent method for assessing the life cycle GHG emissions of goods and services" (BSI, 2008b). **Therefore the primary aim of these methodologies is to provide a consistent approach to GHG reporting.**

In the biofuel-based methodologies there is an emphasis towards regulation of GHG emissions against a benchmark (fossil emissions). The PAS2050 method does not have determined benchmarks; instead the results could be used to assess in-house GHG reduction targets or used for carbon-labelling schemes for consumer awareness (Carbon Trust, 2008).

All the methodologies aim to account for the GHG emissions that arise due to the production of one unit of product. The RED and RTFO methods examine the production of 1 MJ biofuel from a given feedstock, whereas the PAS2050 methods can technically be applied to a unit of any product or service.

It is recommended that biofuel regulation and reporting should consider who is directly responsible for a net change in emissions due to biofuel production (Nuffield Council on Bioethics, 2011). In this case ALCA is the recommended approach as producers have immediate control over any direct emissions they cause during production. A CLCA approach, on the other hand, is better suited for policy analysis, where the overall impact of implanting the biofuel targets is considered in a wider, even global context of producers and consumers (Nuffield Council on Bioethics, 2011). These two approaches will have a different scope, and it is reasonable that they will have different LCA methodological approaches (Aylott et al., 2011). The following section identifies a suitable methodology for performing an ALCA.

10.2. A 'Pure' ALCA

Brander et al. (2009a), and Sanchez et al. (2012) provide slightly differing accounts of the main differences between an ALCA and a CLCA in terms of their application, system boundaries and allocation procedures. There is consensus between the two studies in that:

- An ALCA examines a unit of produce from a specific fuel pathway

- The system boundaries should include direct and indirect sources of GHG emissions that are directly caused by the life cycle of the product studied, specifically those that are controlled by the operator.

Brander et al. (2009a) recommends that in ALCA, upstream GHG emissions are allocated between co-products by either mass, energy content or price. Here, there is no right or wrong option, as the method used should be applicable to the system studied (Kindred et al. 2008a). The pros and cons of these two methods are discussed in **Chapter 2**, **Section 2.5.5**. Allocation by price or energy content is performed in the bioethanol supply chains studied here. Allocation by mass is not applicable, as electricity does not have a mass.

Sanchez et al. (2012), suggests that system expansion can take place in ALCA, though this is not agreed with by Brander et al. (2009a). If system expansion is used in ALCA then it should use average data rather than marginal (Sanchez et al., 2012). Substitution credits may be suitable for co-products where there is an obvious displacement, (e.g. exported electricity obviously displaces grid electricity production) as otherwise the expansion of the system increases the complexity and uncertainty of the analysis and it then begins to resembles a CLCA (Nuffield Council on Bioethics, 2011).

10.3. LCA Methodology in GHG Reporting Methodologies

The following sub-section describes the LCA methodology adopted by the RED, RTFO and PAS2050 GHG reporting methodologies. This includes an assessment of:

- **The scope of the methodologies** – which GHG emission sources are included.
- **Co-product or waste definitions** – definition of co-products, by-products and wastes.
- **Allocation methods** – how upstream GHG emissions are allocated between co-products.
- **Other aspects** – other aspects of GHG reporting contained within the methodologies.

These are described in the following sub-sections.

10.3.1. Scope of the GHG Reporting Methodologies

In LCA, defining the systems boundaries is critical, and has been shown to affect the GHG emission results. Each of these GHG reporting methodologies have taken different approaches when defining the scope of their assessment.

The most defined calculation boundary is seen in the RED. It provides a detailed equation (**Equation 10-1**) that lists the emission sources that need to be accounted for (EC, 2009a). The RTFO and PAS2050 methodologies do not provide such an equation.

The RED assessment spans from crop cultivation and extraction of raw material, to transport, to processing and final distribution and combustion, although the final 'fuel in use' is expected to be zero for biofuels (EC, 2009a). This is despite there being evidence of emissions of CH_4 and N_2O from biofuel combustion (AEA Technology and North Energy Associates, 2010).

In the RED, GHG emissions are credited for carbon sequestration and storage due to agricultural practices, land use change and carbon capture and storage, as well as from emissions saved from electricity co-generation. Cultivation and extraction of raw materials should include emissions from waste disposal and production of inputs such as fertilizers or herbicides. Machinery and equipment manufacture is not included.

Equation 10-1. Greenhouse gas emissions from the production and use of transport fuels, biofuels and bioliquids (EC, 2009a).

$$E = e_{ec} + e_l + e_p + e_{td} + e_u - e_{sca} - e_{ccr} - e_{ee}$$

Where:

E = Total emissions from the use of the fuel

e_{ec} = Emissions from the extraction or cultivation of raw materials

e_l = Annualised emissions from carbon stock changes caused by land-use change

e_p = Emissions from processing

e_{td} = Emissions from transport and distribution

e_u = Emissions from the fuel in use

e_{sca} = Emission saving from soil carbon accumulation via improved agricultural management

e_{ccr} = Emission saving from carbon capture and geological storage

e_{ee} = Emissions saving from excess electricity from cogeneration

The RTFO methodology states that the GHG 'footprint' of a biofuel needs to include the direct, indirect and avoided sources of emissions that arise from its production (Bauen et al., 2008). Direct emissions are those that occur due to direct consumption or use of products, whereas indirect emissions are those that occur elsewhere. Avoided emissions are different to both of these. They account for the emissions that *would have* occurred if the given product or service studied did not exist and did not consume the resources it does. This is based on theoretical 'what if' situations and assumes that some change from the norm has occurred. The RTFO examines changes by applying system expansion to account for the emissions avoided due to a change in production of co-products.

The RTFO suggests that 'minor sources' of GHG emissions are not accounted for (Bauen et al., 2008). This is also suggested in the PAS2050 methodology, by stating that at least 95% of the 'anticipated life GHG emissions' should be accounted for. All the methodologies exclude machinery and other capital good manufacture, however the PAS2050 methodology plans on considering it in future revisions (BSI, 2011).

The RED and PAS2050 methodologies include direct land use change if the land use has been altered since January 2008, or 20 years before the material was used, whichever is the latest. Both the methodologies follow the calculations provided in the IPCC guidelines (Bickel et al., 2006), though the guidelines do not include details for calculating DLUC impacts of straw removal.

10.3.2. Co-Product or Waste?

Chapter 2, **Section 2.5.5** provides an account of the definition of wastes and co-products. This is a vital component of LCA as it affects how GHG emissions are attributed between co-products, or not. Co-products are defined as:

> *"Any of two or more products coming from the same unit process or product system"* (Brander et al. 2009b).

Wastes are defined as:

> "*Any substance or object which the holder discards or intends or is required to discard*" (BSI, 2008a; CEN, 1998; EC, 1996).

Despite these definitions, there is no clear guidance on when a material becomes, and ceases to be, a waste (Brander et al. 2009b). A material could still be classed as a waste despite it having an economic value or not (Brander et al. 2009b). This definition leaves some room for interpretation, which can introduce uncertainty into the LCA methodology.

In an attempt to simplify this issue, the RTFO introduced the concept of the 'by-product' in biofuel GHG emission reporting. A by-product is defined as a co-product that represents less than 15% of the farm or factory gate value (RFA, 2010). When a product is a 'by-product' it is assumed that its consumers have "little influence on the sustainability of the production processes for the original product" and do not need to report on the sustainability of their origin (RFA, 2010). The RTFO therefore performs an economic assessment to determine co-product/by-product status. The Department for Transport has already compiled a list of 'by-products', of which straw is listed (RFA, 2010). In **Chapter 6**, **Section 5.5** it was shown that straw could represent up to 19% of the value of the crop, before baling, although over the last five years has averaged at 8.5%. The PAS2050 does not provide such a distinction point.

The RED does not provide any guidance on what constitutes a co-product or a waste. Instead it makes reference to specific processes that yield more than one product. It specifically states that "agricultural residues" and "residues from processing" are not to be attributed with upstream GHG emissions.

Residues are a special type of product that arise in agricultural systems, as there is an option to harvest them or leave on them onsite. The RTFO assumes that the alternative fate of residues is to leave them on the ground to rot. This is considered to have no impact on GHG emissions, as the residues will decompose releasing only CO_2 in the process, which is regarded as carbon-neutral (Bauen et al., 2008). The RTFO have not included any direct land use change effects of removing the straw from soil on the GHG emission results. The PAS2050 states that "all forms of land use change that result in emissions are to be included" (BSI 2011); so it would appear that DLUC due to straw removal should be included here.

10.3.3. Co-Product Allocation

Allocation is an important issue in LCA, as it can be done different ways, and can greatly affect the results (Gnansounou et al., 2009; Kaufman et al., 2010; Mendoza et al., 2008). Although system expansion is the preferred method to deal with co-products in LCA, it has since been suggested that the method is best suited for consequential LCA. The RED has the view that to account for co-products via system expansion is a method best suitable for policy analysis but maybe not for "regulation of individual economic operators and individual consignments of transport fuels" (EC, 2009a).

The principle described in the original RTFO methodology (Bauen et al., 2008) was that "a biofuel should be attributed with any consequences of a marginal increase in demand". If there is an increase in demand for a biofuel crop then there will be an increase in supply of the associated co-products. The GHG impacts of an increase in supply, and use of these co-products should be included within the boundaries of the biofuel's carbon footprint. Therefore, in the RTFO system expansion is the preferred method of dealing with co-products. This is also true for the PAS2050 methodology. Both the RTFO and PAS2050 state that if substitution-credits are not practicable, then allocation should be performed according to their market value.

All of the methodologies treat exported electricity separately to other co-products. This may be because there is an obvious substitution of grid electricity when electricity is exported. One assumes that the export of 1 MJ of electricity from the biofuel process means that 1 MJ of electricity production no longer needs to be produced by the national grid. The substitution credit represents the avoided GHG emissions from grid-based electricity production.

The methodologies all award different credits to exported electricity. The PAS2050 methodologies assume average electricity production is displaced, whereas the RTFO takes a more consequential approach and assumes marginal production is displaced. The RED awards credits for exported electricity depending on what the exported electricity was produced from. It states in Annex 5, point 16:

> *"Emission saving from excess electricity from cogeneration shall be taken into account in relation to the excess electricity produced by fuel production systems that use cogeneration except where the fuel used for the cogeneration is a co-product other than an agricultural crop residue"*

Therefore exported electricity credits are not awarded if a co-product other than a residue is used for electricity production. In the case study this would apply to lignin in the 2nd generation bioethanol production process.

In the same point, it also states:

> *"The greenhouse gas emission saving associated with that excess electricity shall be taken to be equal to the amount of greenhouse gas that would be emitted when an equal amount of electricity was generated in a power plant using the same fuel as the cogeneration unit."*

This means that the exported electricity credit will be based on if the same fuel was used in a power plant to produce electricity. In the case studies used here, it means that when natural gas is used in the CHP plant the credit will be based on natural gas-generated electricity (which is in fact marginal electricity). If straw is used in the CHP plant then the exported electricity credit will be based avoiding dedicated straw-based electricity. Section 10.5 shows how these credits are calculated.

In summary each methodology treats co-products differently:

- **The original RTFO methodology** – performs system expansion, if not possible then allocation by price. Exported electricity is awarded credits for avoided marginal electricity.
- **The RED/RED-Compliant RTFO methodology** – performs allocation by energy content. Cereal residues and residues from processing are not allocated GHG emissions. Exported electricity is awarded credits depending on the fuel source used in the CHP unit.
- **The PAS2050 methodology** – performs system expansion, if not possible then allocation by price. Exported electricity is awarded credits for avoided average electricity.

10.3.4. Other Aspects of GHG Emission Reporting for Biofuels

There are some other aspects of that are particular to one or just two of the GHG reporting methodologies.

10.3.4.1. Differences in Global Warming Potentials

The global warming potentials of CH_4 and N_2O differ between the methodologies. The PAS2050 uses the updated IPCC 2007 values (CH_4 = 25, N_2O = 298), whereas the RED use less recent numbers (CH_4 = 23, N_2O = 296). This should have a small impact on the results; however the discrepancy should not be occurring, as the GHG methodologies should use the most recent figures (BSI, 2011).

10.3.4.2. RED: Credit for Using Degraded or Contaminated Land

Another aspect only present in the RED is that it awards a credit of 29 g CO_2 eq. MJ biofuel^{-1} if the land used "was not in use for agriculture or any other activity in January 2008" or is classified as "severely degraded land, including such land that was formerly in agricultural use" or "heavily contaminated land". The RED states that areas of high carbon stock and biodiversity should not be used as a biofuel resource.

10.3.4.3. RED: Default Values for Biofuel Supply Chains

The RED provides a list of default values for the GHG emissions from biofuel supply chains "to avoid a disproportionate administrative burden" (EC, 2009a). This is defined in Article 2, point O, as:

"*A value derived from a typical value by the application of pre-determined factors and that may, in circumstances specified in this Directive, be used in place of an actual value.*"

If producers believe they have a lower GHG emission rate to those in the default numbers then they can calculate their actual GHG emissions using the RED calculation (Equation 10-1). It is permitted that if the Member state can demonstrate that the average biofuel and crop cultivation GHG emissions are equal or lower to the default values then all biofuel producers in this state can use the given default values. There is some ambiguity about what is defined as a 'typical', 'average' or 'default' value (Ahlgren et al., 2012). The RED provides both typical and default values, the latter being higher than the former, maybe as some part of a conservative approach. **Table 10-1** lists the default values that are relevant to this study.

Table 10-1. Values for 'typical' and 'default' bioethanol supply chains.

Description	Process Fuel	Typical Value	Default Value
Wheat ethanol	Not specified	32%	16%
Wheat ethanol	Natural gas CHP	53%	47%
Wheat ethanol	Straw CHP	69%	69%
Wheat straw ethanol	Not specified	87%	85%

10.3.5. Summary

A summary of the allocation methods applied in each GHG reporting methodology is provided in **Table 10-2** and **Table 10-3**. There are some instances where there is scope for differences in interpretation in the methods. An example of this is seen in the 1st generation bioethanol production system, where DDGS is a co-product from processing. Literally interpreting the RED methodology, this would not be attributed upstream GHG emissions, although it could be argued that most practitioners will sell DDGS for animal feed, and would regard it as a co-product (Lywood et al., 2009). Therefore, some uncertainty can still be present in how the GHG emissions are allocated, despite the use of GHG reporting methodology, which aims to harmonise calculations (Hennecke et al., 2012).

It is assumed that a 'pure ALCA' could either allocate by price or energy content for all co-products, or apply system expansion to electricity only. This is done by awarding exported electricity with avoided average grid electricity. As a reminder, in 2nd generation bioethanol production it is assumed that the lignin by-product is used as a source of heat and power. The two outputs from 2nd generation bioethanol are bioethanol and exported electricity. In very few instances there is sufficient lignin so that some excess solid fuel is exported (Slade et al., 2009).

Table 10-2. Summary of allocation methods utilised in GHG reporting methodologies for 1st Generation bioethanol production, including options for differences in interpretation.

Methodology	Cultivation Stage	Processing Co-products	
	Wheat and Straw	*Bioethanol & DDGS*	*Electricity*
"Pure ALCA"	Allocated energy content or price	Allocated by energy content or price	Treated as a co-product
"Pure ALCA" (with electricity credits)		Allocated by energy content or price	Credit: Average grid electricity
RED (DDGS not allocated) with natural gas-fired CHP	Everything is allocated to wheat	Everything is allocated to bioethanol	Credit: Electricity generated from natural gas
RED (DDGS not allocated) with straw-fired CHP	Everything is allocated to wheat	Everything is allocated to bioethanol	None
RED (DDGS allocated) with natural gas-fired CHP	Everything is allocated to wheat	Allocated by energy content	Credit: Electricity generated from natural gas
RED (DDGS allocated) with straw-fired CHP	Everything is allocated to wheat	Allocated by energy content	None
PAS2050 (DDGS credited)	Allocated by price.	DDGS awarded credits for animal feed	Credit: Average grid electricity.
PAS2050 (DDGS allocated)	Allocated by price.	Allocated by price.	Credit: Average grid electricity.
Original RTFO	Allocated price.	DDGS awarded credits for animal feed	Credit: Marginal electricity

Table 10-3.Summary of allocation methods utilised in GHG reporting methodologies for 2nd Generation bioethanol production, including options for differences in interpretation.

Methodology	Cultivation Stage	Exported Electricity
2nd Generation	(Wheat and Straw)[b]	
"Pure ALCA"	Allocated by energy content or price	Treated as a co-product
"Pure ALCA" (with electricity credits)		Credit: Average grid electricity.
RED	(No allocation to straw)	Credit: Electricity generated from lignin[c]
PAS2050	(Allocated between wheat grain and straw by price.)	Credit: Average grid electricity.
Original RTFO		Credit: Marginal electricity

a. (Brackets) refer to those instances that only apply to wheat straw

b. There are no co-products from Miscanthus production.

c. If natural gas is used to top-up the fuel supply then the credits will be based on a combination of lignin and natural-gas based electricity

10.4. Parameters for Allocation

Allocation of GHG emissions between wheat, wheat straw, DDGS, bioethanol and electricity in the 1st generation bioethanol production must be determined. Likewise for 2nd generation bioethanol allocation of GHG emissions must occur between bioethanol and exported electricity. Allocation by mass cannot take place as electricity does not have a mass. Instead allocation by price and energy content are discussed in the following sections.

10.4.1. Allocation by Price

Wheat and straw prices are discussed in **Chapter 6**, **Section 6.5**. This section examines DDGS, electricity and bioethanol.

10.4.1.1. DDGS

There are no statistics available on the economic value of DDGS (Yan and Boies, 2013). Values used to be in the range of £68 (Woods and Bauen, 2003) to £80 tonne^{-1} (Mortimer et al., 2004), though a more recent valuation is £197 tonne^{-1} (Hazzeldine et al., 2011). The average price between 2008 and 2011 was £180 (Yan and Boies, 2013).

These values correspond to DDGS that is sold in pelleted form that is dried to 10% moisture content. Therefore, the shadow price of DDGS must be calculated. To recap, the shadow price provides an estimate of the price of a co-product before it is processed into a more valuable produce. For example, DDGS is a product from the liquid waste stream from the fermentation plant, and at the point of separation from the bioethanol distillate it has a moisture content of up to 91% (Bernesson et al., 2006). A great deal of heat and power is required to reduce it to a form in which the DDGS can be pelleted. The value of wet DDGS is hypothesised as being between two and three times less than dried DDGS, based on estimates in the US (FarmweekNow.com, 2013; Lomas and Moyer, n.d.; McDonald, 2011). The cost of wet DDGS is estimated at being between £101 and £131.

10.4.1.2. Electricity

It is assumed that exported electricity is sold to the national grid. The economic value of electricity is based on the current domestic prices from the UK national grid. In March 2013 the average is £0.1266 kWh^{-1} (DECC, 2013a), which is equivalent to £35.167 GJ^{-1}. There has been a steady increase in electricity prices since 2004 (DECC, 2013b), and this is expected to increase over time, tracking the increases in natural gas prices (National Grid, 2011).

10.4.1.3. Bioethanol

Bioethanol prices depend on the feedstock price (Ajanovic and Haas, 2010), however bioethanol is a globally traded commodity. Bioethanol prices are taken from the Chicago Board of Trade (CBOT) futures and options exchange (CME Group, 2013). The prices are provided in US Dollars per gallon, and an exchange rate of £0.6524 $\$^{-1}$ is assumed. In the UK, prices are subject to a fuel duty rate of £0.5795 litre^{-1} and Value Added Tax (VAT) of 20% (HMRC, 2013). Based on this, the current price of bioethanol is £1.196 litre^{-1} (ranging from £1.194 to £1.208).

Following the same method a price of gasoline of £1.298 litre^{-1} is calculated, which is close to the current price of £1.377 litre^{-1} (Europe Energy Portal 2012, assuming a conversion rate of £0.853 Euro^{-1}). This considers a 5p charge for retail and distribution (PetrolPrices.com, 2013). According to the historical trends in the CBOT, in the last 2 years, bioethanol ranges from a minimum of £1.140 litre^{-1} to a maximum of £1.316 litre^{-1}. Based on a density of 0.79 kg litre^{-1} (Elsayed et al., 2003), and is equivalent to £1443 and £1666 tonne^{-1}. No differentiation is made between bioethanol from different sources.

The above is the price for bioethanol at the point of sale. The shadow price of the distillate (**Figure 7-8**) must be calculated. An approximate value could be estimated by calculating the costs of dehydration; however data for this is limited. It is not expected that the cost of dehydration will be as high as distillation, which is approximately \$12 gallon^{-1} for 1st generation bioethanol and \$4 gallon^{-1} for 2nd generation bioethanol (McAloon et al., 2000). This corresponds to approximately £12.50 and £4.2 per tonne bioethanol. The differences in costs are due to the fuel sources used, as 1st generation is assumed to utilise natural gas, whereas 2nd generation is assumed to use the lignin by-product for heat and power. As the energy requirements for dehydration are 7 times lower than distillation (**Figure 7-9** of **Chapter 8**), it is assumed this cost is negligible.

10.4.2. Allocation by Energy Content

As described in **Chapter 5**, **Section 5.4**, the energy content of solid biomass is calculated following the Milne equation, using compositional data from the Phyllis Database (ECN, n.d.). The LHV of bioethanol is 26.8 GJ tonne^{-1} (EUCAR et al., 2006), and as the mass balance of distillate and bioethanol is 1.058 to 1 tonne, only a minor difference in the difference in LHV is assumed.

The energy content of DDGS is calculated based on the moisture contents expected from literature review. This method is somewhat limited at present as the Phyllis Database does not contain specific data on DDGS. Brewing waste is selected as a substitute. A review of a selected number of literature provides a LHV of between 15-18 GJ tonne^{-1} for dry DDGS (Alberichi and Hamelinck, 2010; Gnansounou et al., 2008; Malça and Freire, 2006; Martinez-Hernandez et al., 2013; Punter et al., 2004; Weinberg and Kaltschmitt, 2013), so the value in **Table 10-4** is on the higher end of this scale. The Milne equation estimates a negative energy value for wet DDGS, and this is attributed a LHV of zero. Lignin is assumed to have a LHV of 6.68-9.89 GJ tonne^{-1}, according to a moisture content of 50%-63% (Borrion et al., 2012b).

Table 10-4. Energy content of wheat grain, wheat straw, DDGS and lignin.

Biomass	M. C	Hydrogen	Ash	HHVdaf	HHVar	HHVdry	LHVdry	LHVar
Units	%			MJ kg^{-1}				
Wheat Grain	20	5.05	7.27	19.57	14.52	18.15	17.05	13.15
	14				15.61	18.15	17.05	14.32
Wheat Straw	20	5.5	5.04	19.39	14.73	18.41	17.21	13.28
	15				15.65	18.41	17.21	14.26
DDGS	91	6.87	4.4	23.22	2.00	22.20	20.70	0.00*
	40				13.32	22.20	20.70	11.44
	10				19.98	22.20	20.70	18.39
Lignin	50	5.89	0.62	23.65	8.7	23.5	22.22	6.68
	63				11.75	23.5	22.22	9.89

10.4.3. The Point of Allocation

In ALCA, it is conventional to allocate upstream GHG between co-products when they are created. As stated in **Chapter 5, Section 5.4,** allocation should not include any processes that occur after this to create a more valuable product (Aylott et al., 2012). The RED advises that if any linkage between the co-product's downstream process and the main product's production process exist, the system is considered a refinery and allocation should be made at the point after the final junction (Manninen et al., 2013). Therefore if the same CHP unit is used to provide the heat and power for both bioethanol production and DDGS drying then allocation can take place between the finished products. In this case, the system boundaries would also include DDGS drying (**Figure 10-2**).

Converting the DDGS into a more valuable produce requires a considerable energy input for drying. As the CHP plant could also be scaled to provide sufficient heat and power to dry DDGS to a valuable state. The drying process would, however, increase the heat requirements of the CHP plant by 45 to 117%, meaning that more natural gas is needed to satisfy the plant's energy demand. A larger CHP unit will also lead to a higher net excess electricity output.

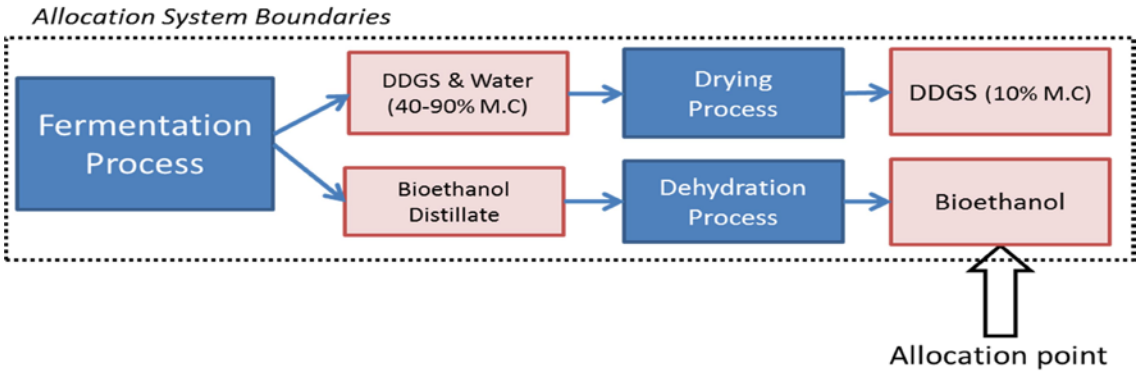

Figure 10-2. Expanded system boundaries to include DDGS drying before allocation between DDGS and bioethanol.

10.4.4. Substitution Credits

In **Chapter 2**, substitution credits defined as GHG credits for avoided production of marginal goods. It is the recommended method for avoiding allocation issues in LCA (CEN, 2006a), although it may be more appropriate for CLCA.

The following co-products are identified in the case studies:

- DDGS
- Electricity

10.4.4.1. DDGS

As DDGS has a calorific value of 11.2 MJ dry tonne^{-1} (to ruminants) and a protein content of 33% (ICCT, 2012), it is assumed to displace marginal sources of animal feed such as soymeal, which is the most widely used source of high-protein animal feed in the EU (Lywood et al., 2009). Substitution credits can then be awarded for displaced soymeal (about 0.3-0.82, average 0.59 kg CO_2 eq. kg DDGS^{-1}) (Bauen et al., 2010; Dehue et al., 2009; Gnansounou et al., 2008; Martinez-Hernandez et al., 2013; Weinberg and Kaltschmitt, 2013; Yan and Boies, 2013). Though the method has been adopted in many LCA studies, it has been deemed too 'simplistic' as it is now understood that the value of the credit is wrought with uncertainty (Yan and Boies, 2013). A recent study models how animal feed displacement by DDGS will depend on the economics, feed characteristics, feed demand and supply and estimates that, at certain prices, a number of products could be displaced (ICCT, 2012). For example, at a price of £180 tonne^{-1} one tonne of DDGS is expected to displace a combination of soy meal (0.37 tonnes), sunflower meal (0.27), feed wheat (0.13 tonnes), maize gluten (0.13 tonnes), barley (0.11 tonnes) and small amounts of other crop meals (ICCT, 2012).

Rape meal and sunflower meal, are co-products from rape and sunflower oil production, therefore their production cannot be avoided (Yan and Boies, 2013). In this case the substitution credit would only account for avoided transport from the major exporters. Soy meal was originally a co-product from soy oil production, but due to increasing demands for animal feed in the agricultural sector, soy meal production is sometimes the main driver for soy bean cultivation (Garnett, 2009). The substitution credit of soy meal ranges between 344-721 kg CO_2 eq. tonne soymeal^{-1}, depending on which marginal oil is displaced by soy oil (Yan and Boies, 2013).

As it is out of the scope of this study to perform a selective LCA of each animal feed, a number of estimates are taken from literature (**Table 10-5**). The substitution ratios described in ICCT (2012) are used to test the sensitivity of the result to different assumptions on animal feed displacement. A nominal transport distance of 100 km (with an empty round trip) is assumed for meal products.

Table 10-5. Substitution credit options for DDGS.

Animal Feed	GHG Emissions (kg CO_2 eq· $tonne^{-1}$)	Reference
Barley	538	North Energy Associates 2006b
Feed Wheat	Based on this study	Based on this study
Soy Meal	533 (344-721)	Average production,(Minimum and maximum) (Yan and Boies, 2013)
Rape/Sunflower/Maize Meal	10.68	Assume transport of 100 km

When DDGS is sold for animal feed it is assumed it has been dried to 10% moisture content. The drying process must be encompassed in the system boundaries of the bioethanol supply chain (Aylott et al., 2012). It is estimated that approximately 0.7 – 1.9 GJ of heat is required to dry DDGS from a moisture content of 40 or 91%, respectively, based on an energy requirement of 2.1 GJ tonne water $evaporated^{-1}$ via batch drying (AEA Technology and North Energy Associates, 2010). It is assumed that the heat can be provided from the CHP boiler but the additional heat demand must be taken into account. An increased heat load will also affect the size of the CHP boiler and the amount of exported electricity.

10.4.4.2. Electricity

The GHG emission factors for exported electricity can be either:

- Average grid
- Marginal grid
- Straw-powered

10.4.4.3. Average and Marginal Grid Electricity

The GHG emissions for average grid and marginal electricity are relatively straight-forward to calculate. Average production GHG emissions range from year to year due to changes in proportion of fuels contributing to the UK energy mix (DEFRA and DECC, 2012). In this study, a 5-year rolling average developed between the years 2006-2010 is used. During this time there was a downward trend in UK electricity GHG emissions. The minimum value is set as the average (145 g CO_2 eq. MJ^{-1}). The maximum is 150 g CO_2 eq. MJ^{-1}.

Marginal emission factors are intended to be used to represent actual changes in GHG emissions due to small changes in electricity consumption (DECC, 2012b). This therefore relies on determining the marginal form of electricity production. This is conventionally assumed to originate from Combined Cycle Gas Turbine (CCGT) plant as they are relatively cheap and quick to build and start up (DECC, 2012b), can respond quickly to fluctuating demand, and are the most efficient mode of producing electricity (IEA, 2010). A typical conversion efficiency for a CCGT plant is 50% (Brown & McLeavey-Reville 2011), though this could reach 64% by 2020 (IEA, 2010). The average efficiency is assumed to be 54.5% (North Energy Associates, 2010).

Average or marginal credits are used in the 'Pure ALCA', PAS2050 and RTFO methodologies. According to the RED, GHG emission credits are not awarded for exported electricity when it is generated from a residue that is not an agricultural residue. Hence, when exported electricity is produced from lignin no credit to the exported electricity is awarded. When it is produced from straw a credit is awarded according to avoided straw-based electricity. The theory is that if one MJ of electricity is exported to the grid then one MJ of straw-based electricity no longer needs to be produced. This logic is somewhat confusing and does not seem to represent current grid electricity. There is only one straw-based power station in the UK; and this will be used as a basis for calculating the credit.

10.4.4.4. The Straw Electricity Plant

As straw-based electricity is only needed in the RED accounting methodology, only the RED calculation rules apply here. This is only used in the 1st generation bioethanol supply chain when straw is used to generate heat and power, and there is excess electricity production.

The emissions are is based on the Elean power station in Ely, UK, which is a straw-based 38 MW plant, which consumes 200,000 tonnes straw $year^{-1}$ (EPR, 2011). It is assumed that the plant receives all straw via road delivery. An average transport distance of 100 km is set as a baseline. Under the RED calculation rules straw is not allocated upstream GHG emissions from cultivation, and it has a "GHG intensity of production" of zero. As the RED excludes machinery manufacture it is assumed that plant construction is not included. Also, as the RED assumes that combustion of biomass has a GHG emission of zero; non-GHG combustion emissions are also excluded.

It is assumed that the conversion efficiency of the power plant is 25%, and 56.7 GJ of natural gas for start-up is required (AEA Technology and North Energy Associates, 2010). The GHG emissions from straw-based electricity are provided in **Table 10-6**.

Table 10-6. GHG emission factors for average and marginal grid electricity and lignin-based electricity (kg CO_2 eq. MJ^{-1})

Electricity Type		Carbon Dioxide	Methane	Nitrous Oxide	Total GHG
UK Average	**Average**	**0.1436**	**0.0001**	**0.0009**	**0.145**
	Minimum	0.1436	0.0001	0.0009	0.145
	Maximum	0.0000	0.0001	0.0017	0.150
Marginal	**Average**	**0.111**	**0.004**	**0.0001**	**0.115**
	Minimum	0.101	0.003	0.0001	0.104
	Maximum	0.121	0.004	0.0001	0.125
Straw	**Average**	**0.003**	**0.000**	**0.006**	**0.010**

10.5. GHG Emissions from 1st Generation Bioethanol Production

This section discusses the results of the 1st generation bioethanol production from wheat grain in the UK. This section is structured into the following:

- **Sources of GHG emissions**
- **Results from the 'Pure ALCA'**
- **Results from the GHG reporting methodologies**
- **Validation of results**

These are discussed in the following sub-sections.

10.5.1. Sources of GHG Emissions

Figure 10-3 shows the sources of GHG emissions in bioethanol production from wheat grain. This shows the result when the GHG emissions are allocated between bioethanol and its co-products DDGS and electricity. DLUC is not yet included.

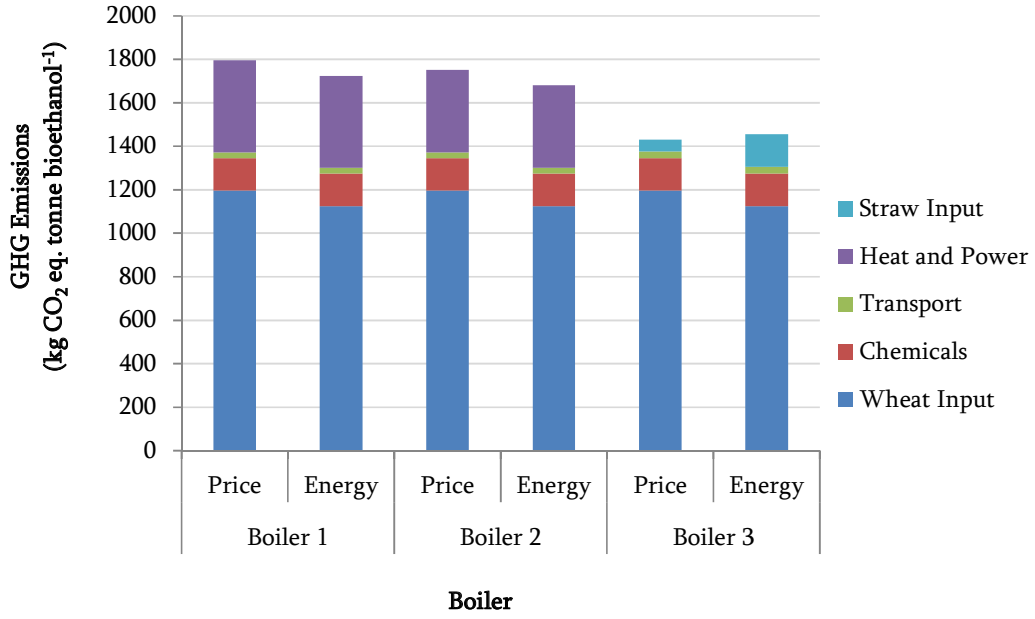

Figure 10-3. Sources of GHG emissions from 1st generation bioethanol production (see Table 8-5 for details on boilers).

The results show that wheat input is the largest source of GHG emissions. This is because wheat is a 'GHG intensive' crop to cultivate. The fuel requirements for conversion (heat, power and chemicals) are also a high source of GHG emissions. Emissions for fuel are higher in boilers 1 and 2 because these utilise natural gas. Boiler 3 uses straw, and it is assumed that the CO_2 emissions from combustion of straw are biogenic, and are hence neutral.

Overall, the GHG emissions per MJ for natural gas are 56.7 g CO_2 eq., whereas for wheat straw it ranges between 0.4 and 25.3 g CO_2 eq. MJ^{-1}, therefore boiler 3 has an overall lower GHG emission rate than boilers 1 and 2. As the differences between boilers 1 and 2 are small only boiler 1 is examined from this point.

Differences can be seen between the GHG reporting methodologies even when examining the same boiler (**Figure 10-4**). A large difference between the ALCA and the methodologies can be seen in how credits are awarded to exported electricity or DDGS. From **Figure 10-4** it is obvious that these credits can deduct significant quantities of GHG's from the total GHG emission result from bioethanol.

The RED awards a much smaller credit to straw-based exported electricity. In the PAS2050 method, emissions from biofuel combustion represent 2% of the total emissions. Emissions from combustion are assumed to be 0 kg CO_2 eq. in the other methodologies. Differences due to different GWP assumptions are also negligible.

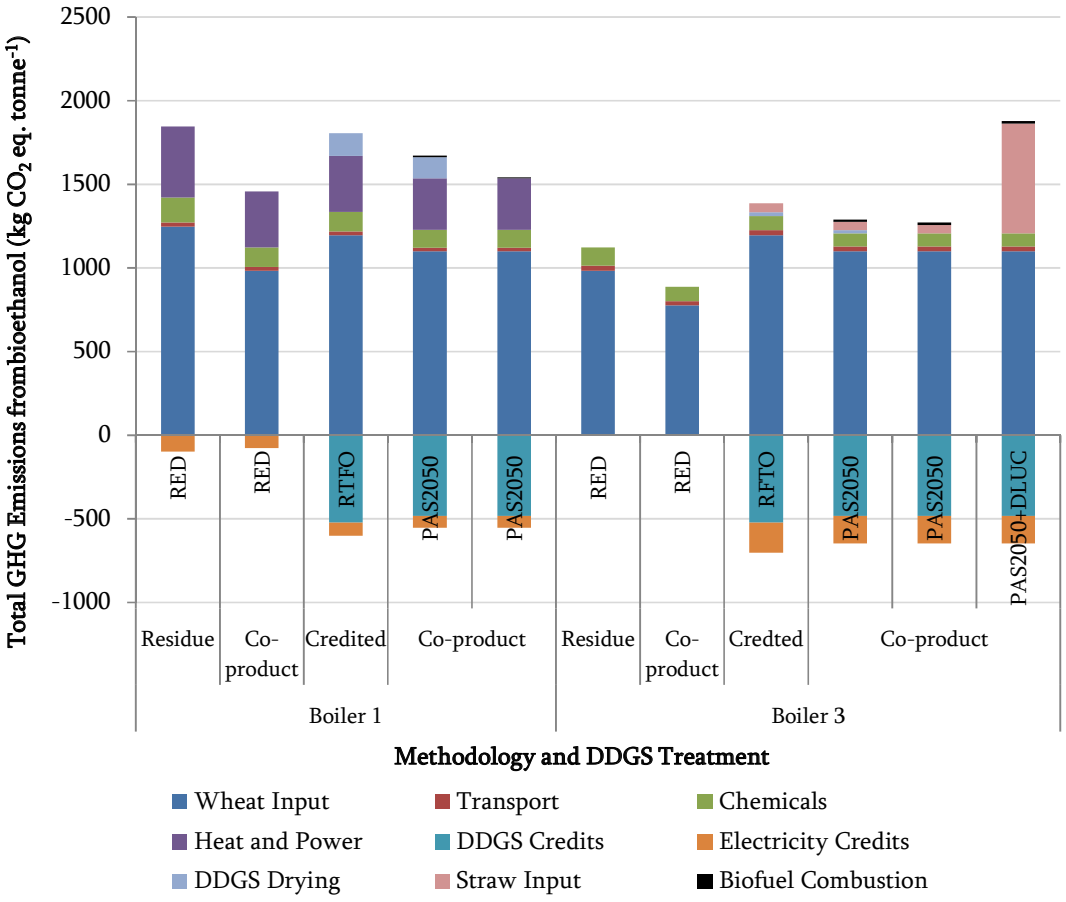

Figure 10-4. Sources of GHG emissions from 1st generation bioethanol production according to the three GHG reporting methodologies, including various options for their interpretation.

10.5.2. Results from the 'Pure ALCA'

Figure 10-5 shows the overall results of the pure ALCA of 1st generation bioethanol production from wheat grain in the UK. The study includes residue removal, allocation between wheat and straw, and between bioethanol, DDGS and electricity according to price (P) and energy content (E). An additional scenario is shown when exported electricity is awarded with average credits for avoided exported electricity (C). The interquartile range with 5% and 95% percentiles are shown. The figure also shows the 35%, 50% and 60% GHG reduction targets as specified by the current UK and European Biofuel policy (EC, 2009a). If the GHG emissions are higher than the target line, it means that the emission saving target has not been met.

The ranges of GHG emission savings are shown in **Table 10-7**, and overall, these range between -32 and 79%. The average savings range between 9 and 57%. The best case scenario is seen when straw is used in the boiler, but this soon becomes the worse-case scenario if it leads to a loss of SOC (DLUC). **Therefore, these results suggest that using straw in the CHP burner can improve the GHG emission savings of bioethanol, but only if it is sustainability removed from soil.** There is some uncertainty in the effect of residue removal, as in some cases not all of the straw from a wheat crop is required to satisfy the heat and power requirements of a crop. In this study it is estimated that between 0 and 3.6 tonnes ha^{-1} of excess straw remains on the site. A sustainable removal limit to avoid negative implications of DLUC must be determined.

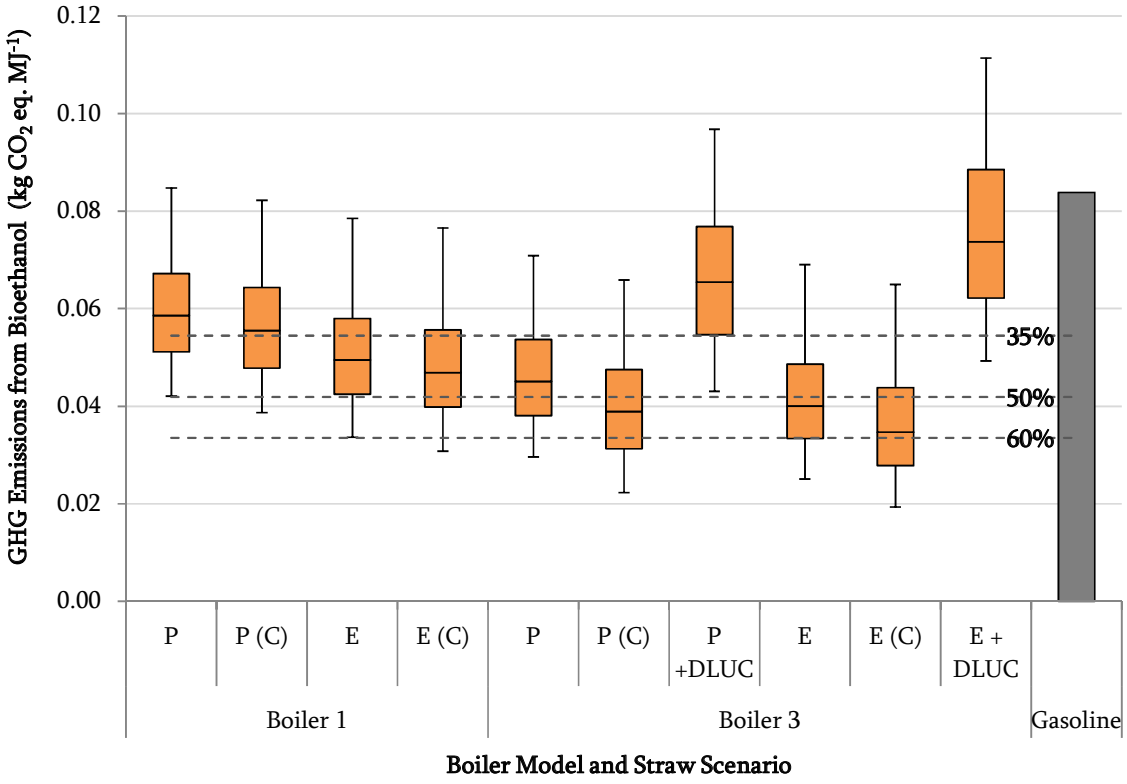

Figure 10-5. Overall GHG emissions per MJ of 1st generation bioethanol when allocated in different ways in a 'pure ALCA'. (P= price, E= energy, C= electricity credited, DLUC = direct land use change).

The results show that wheat-based bioethanol can have a net lower GHG emission rate than conventional gasoline, though not all cases are the 35% emissions saving targets met. For example, with the natural-gas fired boiler 1, the interquartile range lays over and above the 35% savings target when allocated by price. **When DLUC from straw removal is included the 35% savings are completely missed.**

In fewer cases are the 50% and 60% emissions saving targets met. Even when straw is used the GHG emissions are not low enough to satisfy these stricter targets. The natural gas CHP boiler is more efficient than the straw boiler, and can give savings of up to 42%. It appears, from the results, that these stricter saving targets can best be met from utilising a straw-based boiler and wheat with a high yield, low fertiliser requirements, and running a conversion process with a lower energy demand or near-theoretical maximum bioethanol yield.

As the GHG emission saving targets increase the impact of allocation becomes more important. For example, allocating by energy instead of price means that the bioethanol reaches the 60% saving limit in more instances. Allocation by price has lower savings because wheat has a higher relative price compared to straw, therefore accrues a higher proportion of the emissions from cultivation. Also, using a credit for exported electricity also reduces the GHG emissions compared to allocating emissions to electricity. For example, when allocating electricity by energy content, straw-based bioethanol can reach 50% saving targets. When, however a credit is awarded to exported electricity, the interquartile zone falls over the 60% saving marks. **This suggests that the results are highly sensitive to decisions made whether to allocate GHG emissions to electricity or perform system expansion and award credits for avoided electricity production.**

Table 10-7. Results of the LCA showing minimum and maximum % GHG emission savings from 1st generation bioethanol.

	Boiler 1				Boiler 3					
	P	P(C)	E	E(C)	P	P(C)	P(DLUC)	E	E(C)	E(DLUC)
Average	28	32	39	42	44	51	19	50	56	8
Max	50	54	61	65	65	73	48	70	79	41
Min	2	5	10	12	17	25	-15	20	26	-32

10.5.3. Results from the GHG Reporting Methodologies

Figure 10-6 shows the GHG emission results that are generated by specific GHG reporting rules in the RED, RTFO and PAS2050. It also shows the impact of differences in the interpretation of the rules. The interquartile range with 5% and 95% percentiles are shown, along with the GHG emission limits for reaching the 35%, 50% and 60% saving targets. These results are all generated using the same model dataset.

The results show large differences in the results due to the different accounting rules provided in each GHG reporting methodology.

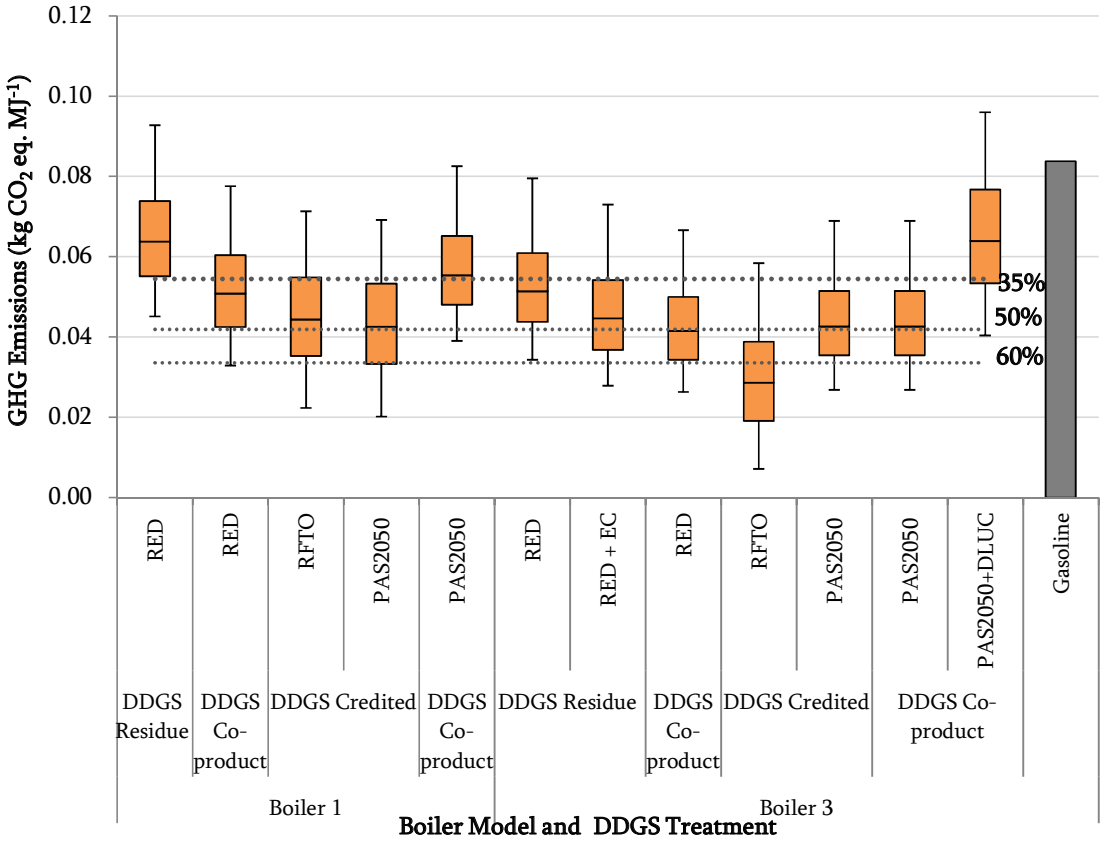

Figure 10-6. Overall GHG emissions per MJ of 1st generation bioethanol when calculated according to different GHG reporting methodologies. The RED +EC category is the RED methodology with an electricity credit awarded when straw is used.

Overall, the GHG emission savings range between -14 and 94%, although on average they range between 21 and 69%. Therefore, compared to the pure ALCA both the overall and average GHG emission savings generated by the GHG reporting methodologies show a larger range. The observed range may be due to the introduction of substitution credits for DDGS, which were not included in the pure ALCA.

There are several key differences in the results which are due to specific methodological differences. These are discussed below.

The RED does not award the same credits to exported electricity when straw is used in the CHP plant as compared to natural gas. If they did award marginal credits to exported electricity the GHG emission savings would be almost 10% higher (see RED+EC for 'electricity credit').

The RTFO applies marginal credits for exported electricity and the PAS2050 applies average credits. Average credits are higher than marginal credit because average credits include average generation of UK electricity, of which some originates from coal (MacLeay et al., 2010). Marginal credits are based on CCGT natural gas which has a higher efficiency (IEA, 2010).

Each methodology treats DDGS differently, or there can be different interpretations on how to calculate this. In the RTFO methodology DDGS is always credited with avoided GHG emissions from displacing animal feed. Due to differences in interpretation, the PAS205 methodology could either apply credits to DDGS or allocate by price. Likewise the RED could either ignore DDGS or allocate it by energy content. The option of whether to allocate or award credits for avoided animal feed has very large impacts on the results.

Another reason why allocating DDGS by price or energy content has a smaller impact on the results is because it is assumed that allocation occurs when DDGS is in a wet and invaluable state. There can be some uncertainty in this. Conventionally, allocation should occur when the stream of DDGS leaves the distillation process with a moisture content of 91 to 41% (Aylott et al., 2012). DDGS has a lower energy content and price than after it is processed for selling as an animal feed in pellet form and a moisture content of 10%. As the CHP plant could also be scaled to provide sufficient heat and power to dry DDGS to a valuable state, the drying process could be 'linked' with the bioethanol process (**Figure 10-2**). The drying process would, however, increase the heat requirements of the CHP plant by up to 117%, meaning that more natural gas is needed to satisfy the plant's energy demand, and more excess electricity is exported. **Figure 10-7** shows how allocating before or after DDGS drying affects the GHG emission result.

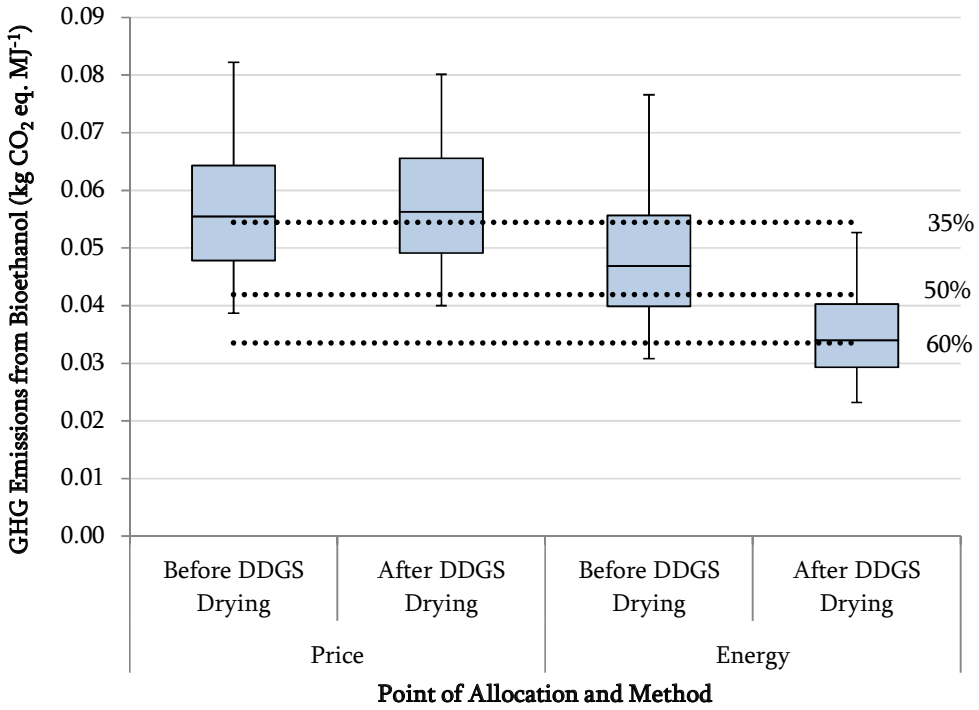

Figure 10-7. Effect of allocating before or after DDGS drying. This includes increased energy requirements for drying and increased electricity credits from a larger CHP plant.

The results show that small differences are seen when allocating by price, which may be because the price difference between bioethanol and DDGS is large in either case. In contrast, the GHG emission results are reduced when allocating by energy content. This is because the relative energy content of DDGS is increased from 0-11 to 18 GJ tonne^{-1} after drying. Despite the GHG emissions from the whole process being increased by drying DDGS, a larger heating requirement means that the CHP unit used is larger, and there is greater net excess electricity from the process. Therefore the GHG emission savings of bioethanol are increased from an average of 45% to 66%: due to a combined effect of the electricity credit and a greater allocation to DDGS.

10.5.4. Validation of Results

The results from this study are compared with 6 other studies in the literature using either price or energy allocation, and utilising either natural gas or straw-fired CHP. The results appear to be consistent with literature (**Figure 10-8**). One exception is seen with natural gas CHP with a higher estimate provided in Martinez-Hernandez et al. (2013), although it is difficult to determine why this is. Also, where straw is used as a fuel source the results estimate a greater GHG emission value per MJ bioethanol than other studies. This may be due to lower prices estimated in Mortimer et al. (2004), a high fertiliser penalty for straw removal in Punter et al. (2004) and the Biograce model (Biograce.net, 2012) applies the RED methodology, where straw is not allocated GHG emissions from cultivation.

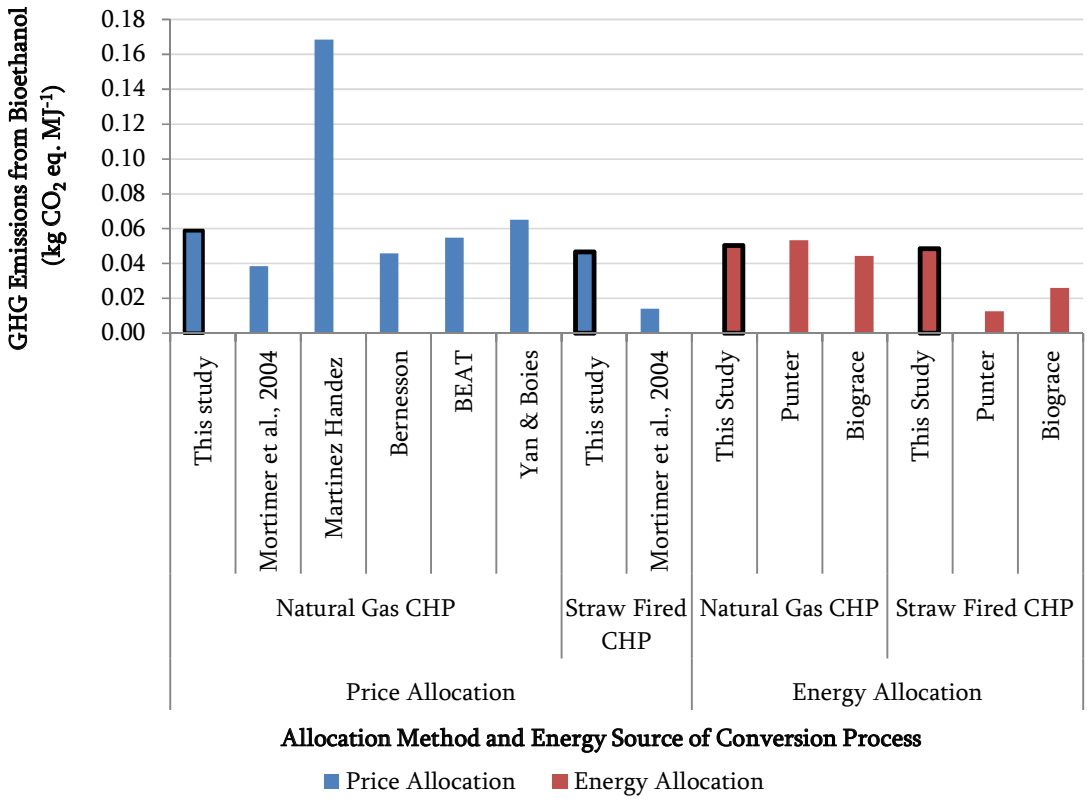

Figure 10-8. Comparing the results of this study with other similar studies from literature.

10.6. GHG Emissions from 2nd Generation Bioethanol Production

This section discusses the results of the ALCA of 2nd generation bioethanol production from wheat straw and Miscanthus in the UK. The results are structured as following:

- **Sources of GHG emissions**
- **Results from the 'Pure ALCA'**
- **Results from the GHG reporting methodologies**
- **Validation of results**

10.6.1. Sources of GHG Emissions

The main sources of GHG emissions of 1st and 2nd generation bioethanol production are shown in **Figure 10-9**. Differences can be seen between the 1st and 2nd generation processes. The main difference is that the biomass input is not the main contributor to overall GHG emissions in 2nd generation bioethanol. A large proportion of GHG emissions from wheat cultivation are allocated to grain and Miscanthus is a very low GHG-intensive crop. Industrial-standard Miscanthus has a lower GHG emission result than the literature-based autumn and spring scenarios. The autumn and industrial Miscanthus scenarios are examined from this point.

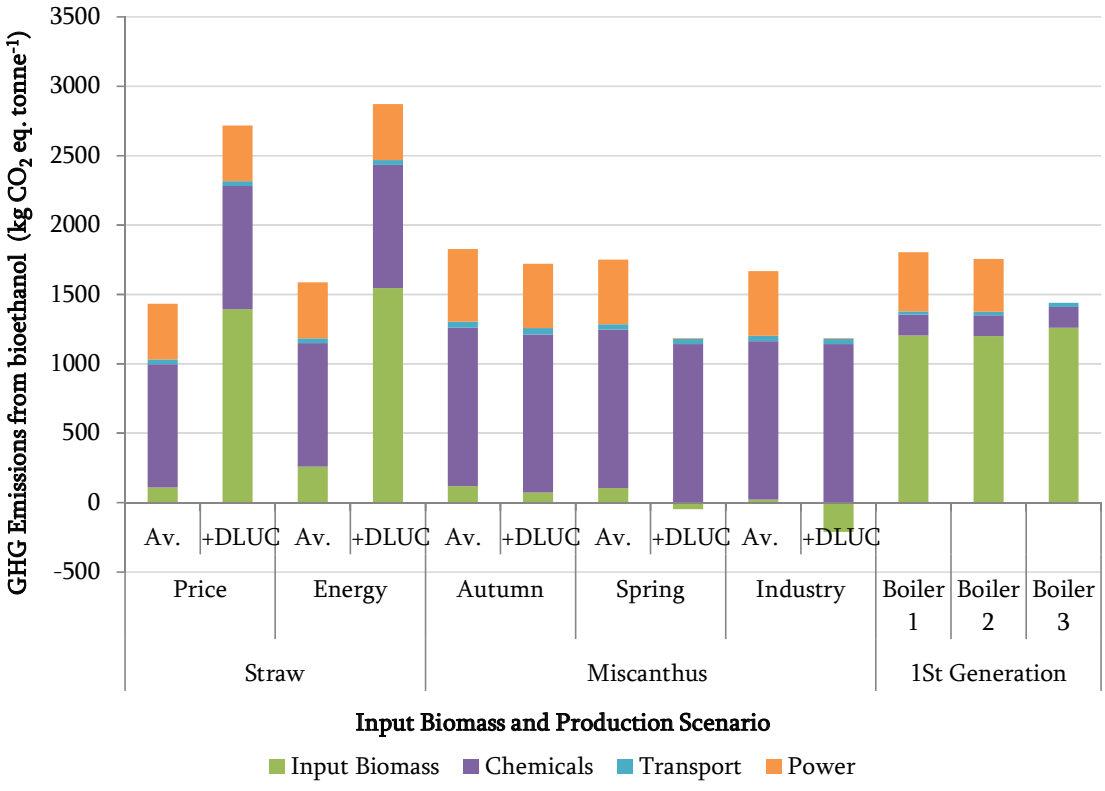

Figure 10-9. Sources of GHG emissions in 2nd generation bioethanol production.

Another main difference between the two bioethanol types is seen in the GHG emissions from chemical and power requirements. In 2nd generation bioethanol, it is assumed that the power requirements are satisfied by the lignin waste stream from the input biomass. Instead the majority of GHG emissions originate from chemicals used in pre-treatment, saccharification and fermentation, including off-site energy requirements for producing enzymes. The GHG emissions from transport are small.

Applying allocation by price and energy content creates differences in the results. Allocation by price means the GHG emissions per tonne bioethanol are smaller. **Therefore allocation by energy content favours 1st generation bioethanol and allocation by price favours 2nd generation.**

When examining the sources of GHG emissions between the different GHG reporting methodologies (**Figure 10-10**), some differences can be seen between these and the pure ALCA shown in **Figure 10-9**. These differences may be because the pure ALCA did not perform system expansion for exported electricity, rather treated it as a co-product. From **Figure 10-10** it is obvious that these can deduct significant quantities of GHG's from the total GHG emission result from bioethanol. In the RED electricity credits are not awarded when lignin is used. The RED does not attribute GHG emissions from wheat cultivation to straw and only includes baling and transport; therefore this impact is smaller in the RED compared to the RTFO and PAS2050, which allocate by price. This makes a small difference to the results.

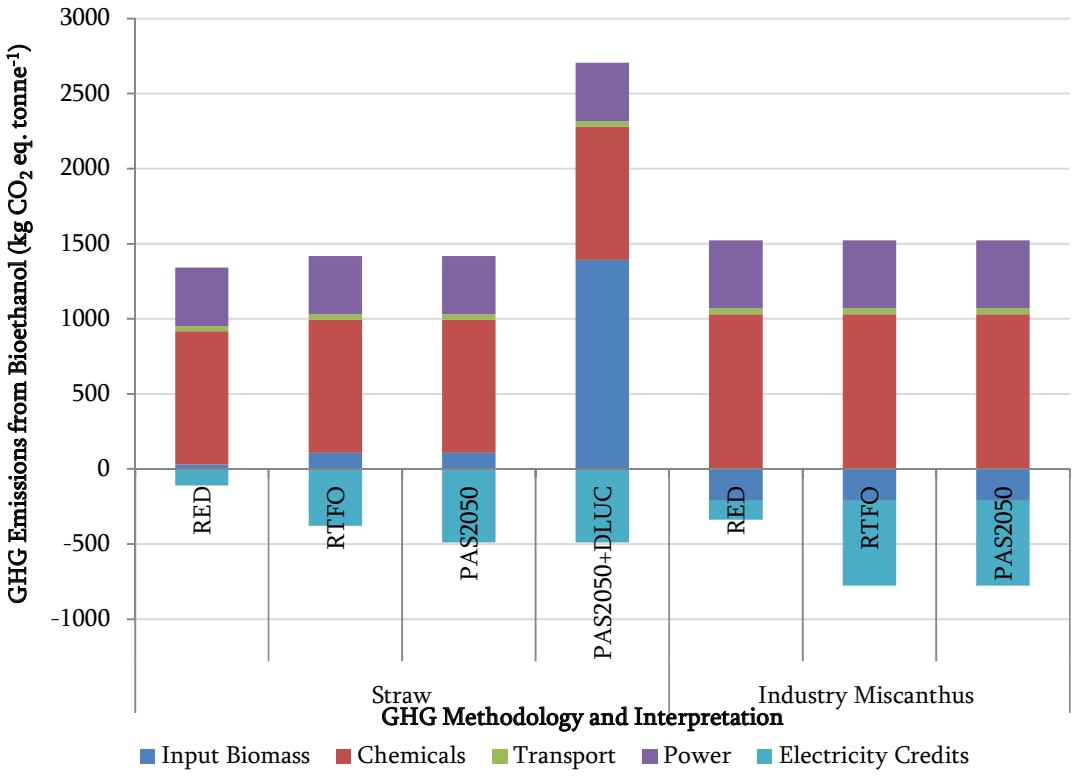

Figure 10-10. Sources of GHG emissions from 2nd generation bioethanol production according to the three GHG reporting methodologies studied.

10.6.2. Results from the 'Pure ALCA'

Figure 10-11 shows the overall results of the pure ALCA of 2st generation bioethanol production from wheat grain in the UK. This includes residue removal, allocation between wheat and straw, as well as between bioethanol, DDGS and electricity according to price and energy content. The interquartile range with 5% and 95% percentiles is shown.

Overall, the GHG emissions savings for 2nd generation bioethanol from wheat straw in the UK range between -55 and 61% (**Table 10-8**). The average savings range from -10 to 43%. The worse-case scenario is seen when using wheat straw for heat and power and when it is assumed that straw removal from soil has caused losses of SOC. The best case scenario is to utilise spring-harvested Miscanthus, with DLUC accounted for. Also, wheat straw, when allocated by price can also give high GHG savings.

The 35%, 50% and 60% GHG reduction targets are also shown, and the results suggest that wheat straw-based bioethanol can be expected to fulfil the current 35% GHG emission saving target, however if DLUC is included in the assessment then this is not the case. The results suggest that Miscanthus-based bioethanol can reach these targets including or excluding DLUC.

The results suggest that industrial Miscanthus has a greater chance of fulfilling 50% emission saving targets; however some intervention may be required for 60% savings to be achieved. This would involve selecting sites or conditions that give the highest yields. Other options for reducing GHG emissions from 2nd generation bioethanol from both straw and Miscanthus could be to use renewable sources of energy to manufacture enzymes and chemicals required in the processing.

Table 10-8. GHG emission savings from wheat straw and Miscanthus-based bioethanol (%).

		Straw		Miscanthus			
Allocation Method				Autumn		Industry	
		No DLUC	+DLUC	No DLUC	+DLUC	No DLUC	+DLUC
Price	**Average**	62	29	53	55	61	72
	Max	61	51	52	54	58	69
	Min	22	5	3	7	17	25
Energy Content	**Average**	57	27	53	56	62	72
	Max	56	48	53	55	59	68
	Min	17	2	5	9	19	25

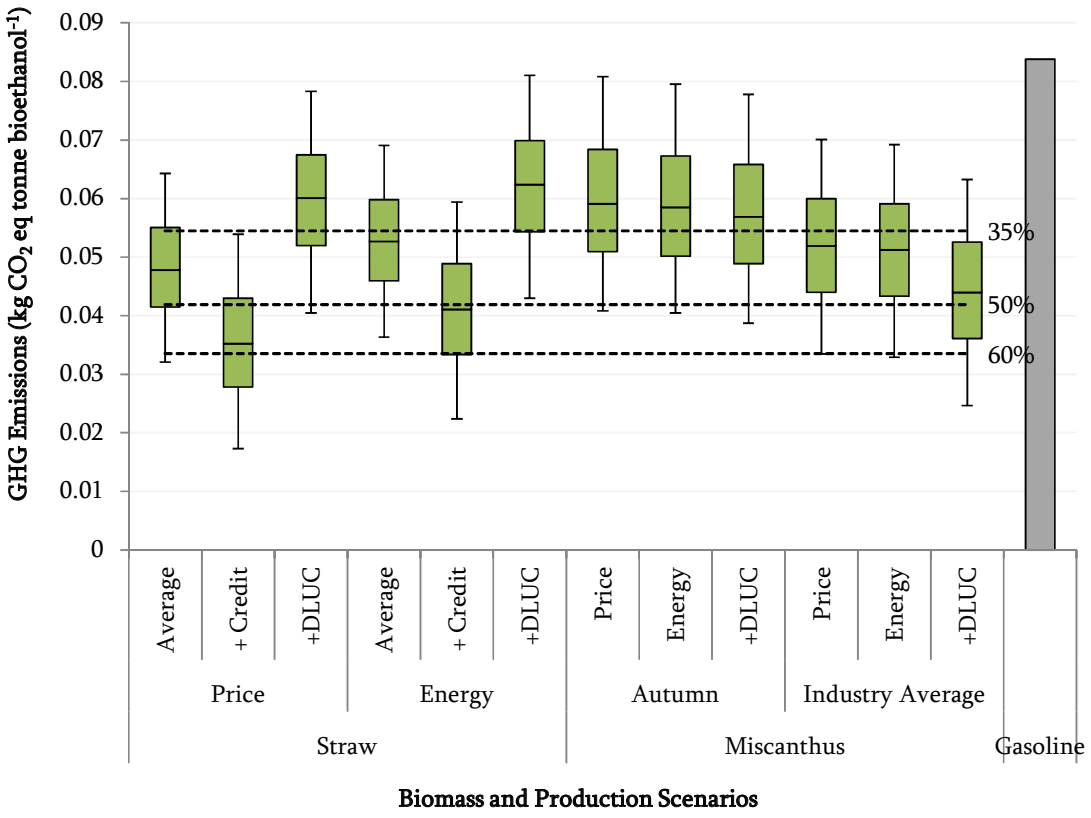

Figure 10-11. Overall GHG emissions per MJ of 2nd generation bioethanol from various feedstocks compared to gasoline and GHG emission saving targets.

10.6.3. Results from the GHG Reporting Methodologies

Figure 10-12 shows the GHG emission results that are generated by the RED, RTFO and PAS2050 GHG reporting methodologies. The interquartile range with 5% and 95% percentiles are shown, along with the GHG emission limits for reaching the 35%, 50% and 60% saving targets. These results are all generated using the same model dataset.

When calculated according to the GHG reporting methodologies, the GHG emissions savings for 2nd generation bioethanol range between -47 and 94% (**Table 10-8**). The average savings range from 1 to 66%. Therefore, compared to the pure ALCA-case the range of results has increased when calculated according to the current GHG reporting methodologies. This also occurred in the 1st generation bioethanol case study and was caused by differences in animal food and electricity credits. Here the difference is mainly due to electricity credits, as none of the co-products can displace animal feed.

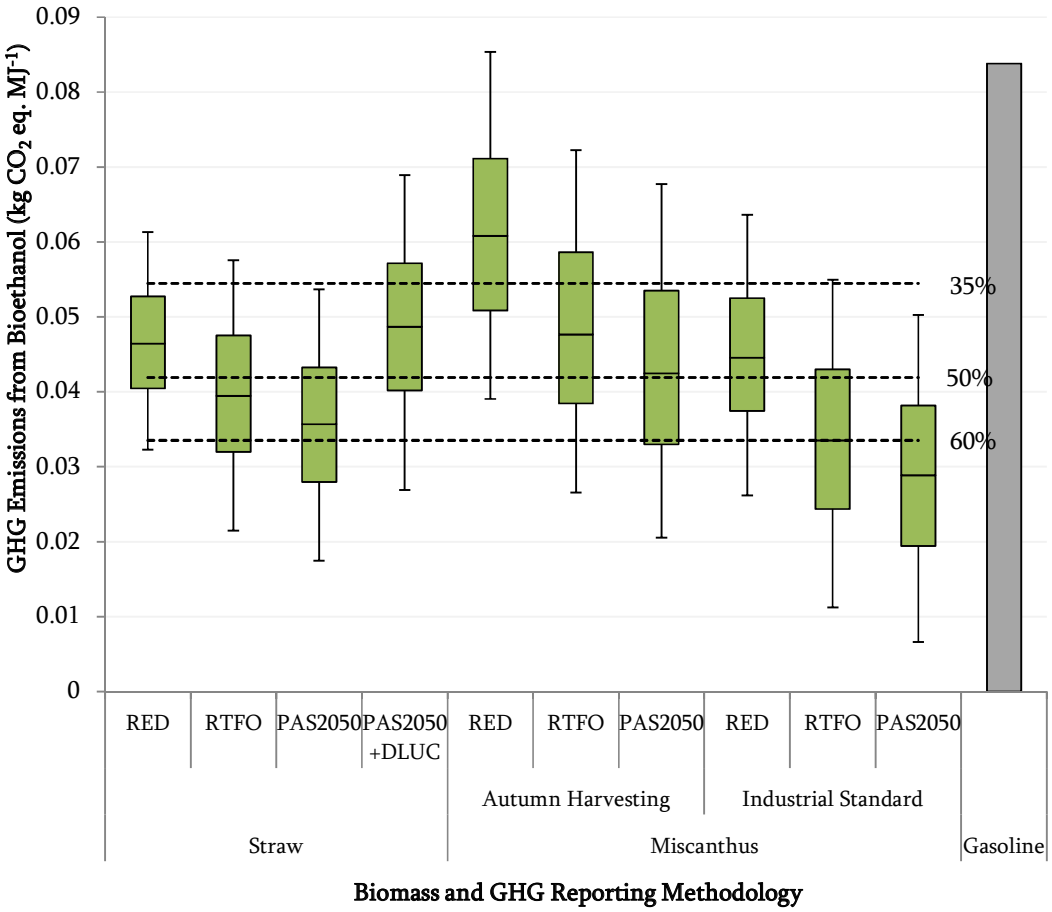

Figure 10-12. Overall GHG emissions per MJ of 2nd generation bioethanol when calculated according to different GHG reporting methodologies.

The main cause of differences in the results is from the electricity credits awarded to exported electricity. The credits affect 2nd generation bioethanol's ability to reach GHG emission saving targets. Under the RED calculation rules, straw based bioethanol makes, on average, a 40% saving, whereas under PAS2050 it is almost 60%. In either case it is exactly the same material produced under the same conditions, yet due to accounting methodology alone the GHG emission savings have shifted by 20%. The difference is due to the lack of electricity credits awarded to renewable-based electricity generated in the CHP plant when calculated under the RED. These differences were not seen in the pure ALCA-case, where the results were more consistent, with small changes caused by different allocation methods.

This pattern can also be seen in autumn and industry-standard Miscanthus, with the RED providing the largest estimate. This is despite the RED not allocating GHG emission to straw. Therefore, the effect of the electricity credit has a greater effect on the results than allocation.

10.6.4. Validation of Results

In the literature, there are only two studies that are sufficiently transparent in order to compare these results with those of literature. Borrion et al. (2012) calculated a total GHG emission of 4203.6 kg CO_2 eq. tonne bioethanol^{-1} from wheat straw. The study did not consider DLUC or range data and allocated by price. According to the pure ALCA, this study gives a total GHG emission of 688 to 1316 kg CO_2 eq. tonne bioethanol^{-1}, and has appeared to underestimate the GHG emissions.

Borrion et al. (2012) estimated that 25% of GHG emissions per tonne of bioethanol originate from cultivation, and the remaining from conversion. Examining the results in close detail it appears that the GHG emissions from input biomass is larger in Borrion et al. (2012) by an average of 9 times of this study, and conversion emissions are three times-higher. A closer examination finds that Borrion et al. (2012) has assumed very low yields of wheat per hectare (5.3 tonnes ha^{-1}), has assumed allocation by mass, and assumes an 'old fashioned' approach of using a natural gas boiler for heat, and electricity is produced from coal and oil, though they acknowledge that lignin could be used. Hence, the results of the study would be expected to be different to that performed here.

In comparison, Slade et al. (2009) reported an average GHG emission of 15-33 kg CO_2 eq. GJ^{-1}, compared with 25-49 from this study. In Slade et al. (2009), lignin is used for all heat demands, except a small import of electricity.

10.7. Sensitivity Analyses

This sub-section details the sensitivity of the results of 1st and 2nd generation bioethanol production according to some main contributors to the net GHG emissions. This is performed for 1st and 2nd generation bioethanol production by changing the average value for each parameter by 10% increments and decrements until the parameter is doubled or removed from the analysis.

Figure 10-13 shows the results from the 1st generation bioethanol production process. The most sensitive parameter is the biomass input, then the power source used. In terms of methodological aspects, **Figure 10-13** shows that the results are only sensitive to the price of bioethanol when it is low; otherwise it shows that the results are highly sensitive to GHG emissions awarded to electricity credits.

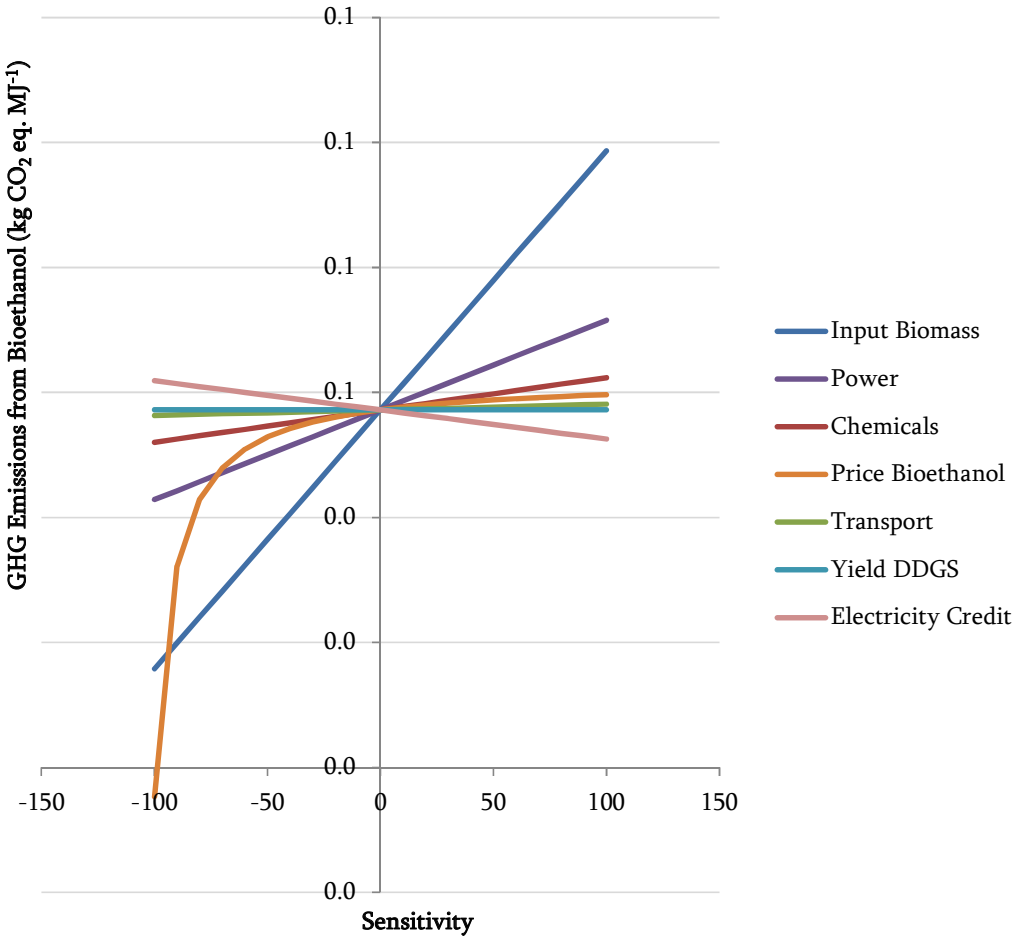

Figure 10-13. Sensitivity analysis of net GHG emissions from 2nd generation Bioethanol according to some key parameters.

Figure 10-14 shows the same analysis for the 2nd generation bioethanol production process. It shows that the net results are highly sensitive to the chemical requirements for conversion and the electricity credits awarded to exported electricity. Interestingly, the GHG emissions are not as sensitive to input biomass, as seen in the 1st generation bioethanol production. This may be because the chemical requirements currently dominate the sources of GHG emissions in the 2nd generation process (**Figure 10-10**).

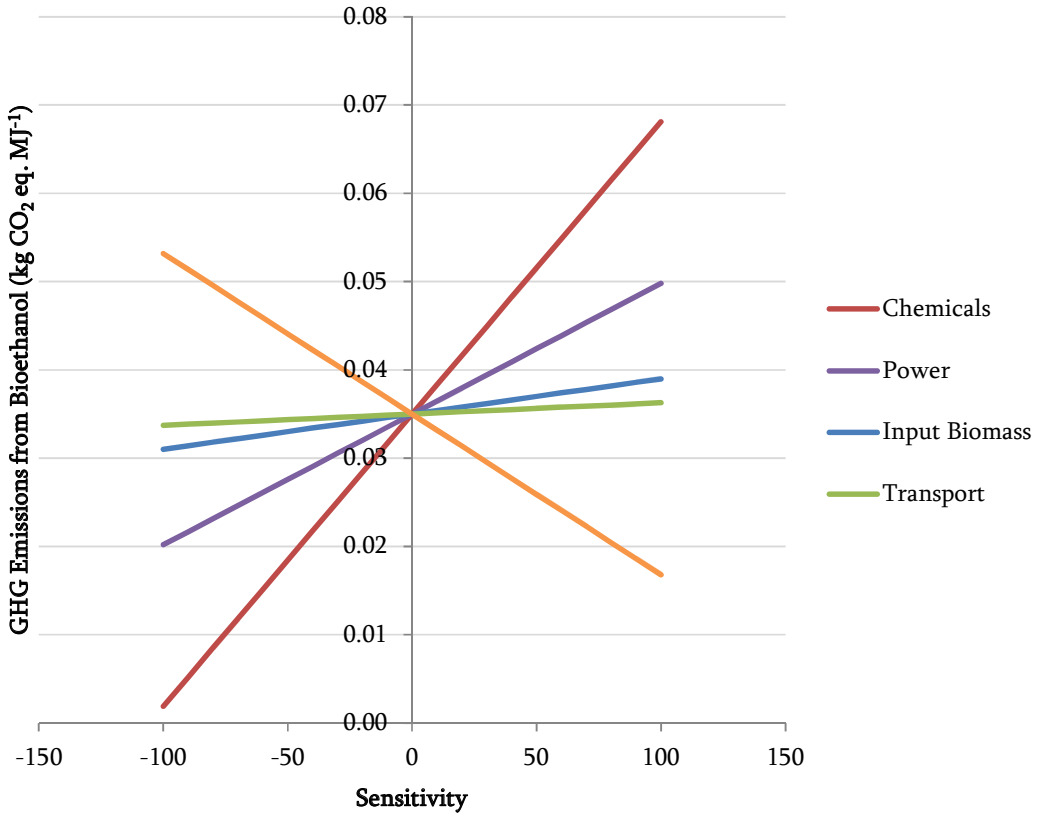

Figure 10-14. Sensitivity analysis of net GHG emissions from 2nd generation Bioethanol according to some key parameters.

10.8. Discussion

It is difficult to define the LCA calculation rules within the GHG reporting methodologies as some aspects are vague and open to interpretation. The methodologies lack both definitions and demonstrations to how different products should be regarded in calculations. Different interpretations may affect the calculation methods and subsequently, the results. The interpretations made in this study are summarised in **Table 10-2** and **Table 10-3**. There are some cases when it is not clear whether a co-product is allocated GHG emissions, awarded credits, or ignored.

The PAS2050 methodology could be considered to be the simplest method: requiring that during the production, use and disposal of a product or service, all sources of emissions that make a 'material contribution' should be accounted for. This may require more guidance however, for reporting specifically on biofuels. The equation provided in Annex V Section C of the RED was specifically written for this but does not provide enough details for the reporting calculations to avoid differences in interpretation, and hence is not practical for use in regulation.

The following sub-section further discusses these differences between the GHG reporting methodologies, and also how they differ from the 'pure ALCA' identified in Section 10.3. This discussion section is broken down into some main sections entitled:

- **The pure ALCA and the GHG reporting methodologies**
  - Co-product and waste definition
  - Co-product allocation
- **Uncertainty in GHG emission accounting for biofuels**
- **Causes of variation in the overall results**

These are discussed in the following sub-sections.

10.8.1. The Pure ALCA and the GHG Reporting Methodologies

The analyses performed in this chapter have shown that there is a substantial difference between what could be described as a 'pure ALCA' and the methodologies employed in the RED, RTFO and PAS2050 GHG emission reporting methodologies. These differences have caused some unexpected results in the GHG reporting methodologies. These key differences are seen in how co-products are identified and treated in the GHG emission calculations.

10.8.1.1. Co-products and Waste Definition

In the pure ALCA, co-products were identified as any product that has a monetary value: such as wheat straw, DDGS, exported electricity and in some cases where it is produced in excess: lignin. All of these products have monetary value however they are treated completely differently in each GHG reporting methodology.

The RED applies specific rules for specific feedstocks. Agricultural residues and residues from processing are not attributed with upstream GHG emissions. This is an arbitrary rule that does not comply with the ISO Standards.

The RTFO suggests that co-products will only be attributed GHG emissions from co-products if they represent above 15% of the monetary value of the crop (RFA, 2010). If wheat straw reaches the threshold price, it is not clear whether it will no longer be classified as a true co-product, as it is already on the designated list of 'by-products'. If changeable, it is possible that at a given price of straw the GHG emissions would change from zero to approximately 346 kg CO_2 eq. $tonne^{-1}$, and the resulting GHG emission savings would change from an average of 45% to 31%. Hence, decision of whether to allocate or not has causes a shift change of 14% in the GHG emission savings of 2nd generation bioethanol. This may represent a risk to bioethanol producers. It is also strange that the status of co-product and waste is determined by price, however under the RED and the RED-compliant RTFO co-products are allocated by energy content. This is a somewhat inconsistent approach.

10.8.1.2. Co-Product Allocation and Substitution Credits

There are differences in how co-products are treated in the GHG emission calculations in the GHG reporting methodologies. Awarding substitution credits gives a higher GHG emission saving than allocation. The GHG emission savings from displacing animal feed and grid-electricity are high; therefore the credits awarded to bioethanol are also high.

The RED also treats exported electricity differently according to its source. The current GHG rules mean that two processes will be awarded different credits depending on their fuel source. For example:

1. **Process A:** produces 1st generation bioethanol utilising a natural gas-fired CHP boiler. There is an excess of 1 GJ electricity from the process, which is exported to the national grid. The bioethanol is awarded 145 kg CO_2 eq. as a credit.
2. **Process B:** produces 1st generation bioethanol utilising a straw-fired CHP boiler. There is an excess of 1 GJ electricity from the process, which is exported to the national grid. The bioethanol is awarded 10 kg CO_2 eq. as a credit.

This is evidence that the current RED methodology penalises the use of renewable energy. In the above example, Process B is not awarded with equivalent GHG credits when renewable energy is used to power the process. Renewable sources can help decrease the GHG emissions from the conversion phase. The full benefit of this saving is not realised if the exported electricity does not receive the same credit. The results show that the GHG emission savings from both 1st and 2nd generation bioethanol are lower than expected as a result of this. Also, the results have shown to be highly sensitive to the GHG emissions awarded in electricity credits.

There is no apparent reason behind this rule in the RED. One suggestion may be because it has actually adopted a consequential-style approach to calculating the GHG emission credit from using renewable energy. It may where the RED has considered the reference system of the biomass involved. A reference system considers the alternative fate of the biomass that no longer occurs if it is used for bioethanol production. For example, it may consider:

1. **Process A:** produces 1st generation bioethanol utilising a natural gas-fired CHP boiler. The straw is used for electricity production.
2. **Process B:** produces 1st generation bioethanol utilising a straw-fired CHP boiler. The straw can no longer be used for electricity production; which has displaced 1 GJ of straw-based electricity.

This is just one possible explanation to why the rule applies in the RED, however even this would not be the correct way of accounting for avoided straw electricity. In this case the straw removed from the site would potentially produce more than 1 GJ of electricity if it was combusted in a dedicated straw plant. It seems more reasonable to assume that when 1 GJ of electricity is exported to the national grid, then 1 GJ of average grid electricity has been displaced, or even 1 GJ of marginal electricity has been displaced. It is argued that marginal credits are only applicable to CLCA (Brander et al. 2009a).

10.8.2. Consequential Aspects in GHG Reporting Methodologies

The analysis of the GHG reporting methodologies has indicated that there are some consequential aspects in them. These are:

- **RED** – substitution credits are applied to exported electricity. This is based on the same fuel being used in a dedicated electricity plant.
- **RTFO** – Applies substitution credits to exported electricity based on marginal electricity production. Applies substitution credits to DDGS (or does not, depending on interpretation).
- **PAS2050** - Applies substitution credits to exported electricity based on average electricity production. Recommends the use of substitution credits for other co-products but does not provide guidance on this.

The RTFO methodology was originally described as 'partially consequential' (Brander et al. 2009a); however has recently been adapted to follow the RED methodology. Neither the PAS2050 nor RED can be described as being 100% ALCA, mainly due to how exported electricity is credited, and the PAS2050 recommends that substitution credits are used where 'practicable' (BSI, 2008b). In the RED, it may be possible that consequential-style methodologies are adopted because the RED is part of an EU renewable energy policy. It is suggested that CLCA is suited for policy analysis (Brander et al. 2009a; Sanchez et al. 2012), therefore **the RED may have confused this approach when designing their calculation methodologies for regulation.**

A CLCA has a different goal and scope to an ALCA study (Brander et al. 2009a; Sanchez et al. 2012; Nuffield Council on Bioethics 2011). Therefore, introducing consequential aspects to an attributional study may mean that the original goal and scope is not being fulfilled. For example, an ALCA aims to account for the GHG emissions that are directly attributed to a single unit of bioethanol. By introducing substitution credits it means that the consequences of bioethanol production on the wider market are being accounted for, which not only complicates the analysis, but means it loses its main goal.

10.8.3. Uncertainty in LCA Methodology

There are several aspects of uncertainty in GHG reporting, and these are discussed here.

There uncertainty in the definition of co-products, by-products and 'wastes', and this can have impact on the final results. The examples of this can be seen in the results regarding wheat straw and DDGS when accounting for GHG emissions following the RED.

There is uncertainty in the allocation method to be adopted. Allocating GHG emissions can be performed in a number of ways, and this has been demonstrated to affect the results. Also, deciding the value of a substitution credit can be wrought with uncertainty (Yan and Boies, 2013). For this reason, system expansion is generally avoided in ALCA. According to the substitution ratios given in Yan & Boies (2013) the GHG emission credit for displaced animal feed ranges between 248 and 688 kg CO_2 eq. tonne $DDGS^{-1}$.

There is uncertainty over the point at which allocation can take place. Conventionally, allocation should occur between co-products at the point at which they are created. There can be some uncertainty in this, because often products are not attributed a price until they are processed and ready to be sold. Here some assumptions were made to estimate the 'shadow prices' of co-products, which involved estimating the cost of processing, such as baling in straw, or drying in DDGS, and deducted it from the market value. In the case of this study, expanding the system boundaries to include DDGS drying had a very large impact on the results, but only when allocating by energy content.

10.8.4. Default figures in the RED

The RED provides some default figures for GHG reporting however they are neither detailed nor referenced, and are not transparent. It is not possible to determine why the values presented in the RED represent higher savings than the average results of this study, when calculated according to the RED calculation methodologies. Higher GHG emission savings are estimated using the RTFO and PAS2050 methodologies. It is interesting to note that the 'unspecified' wheat bioethanol RED default value does not satisfy the 35% emission saving target (**Table 10-9**).

Table 10-9. RED values compared to the results of this study.

Description	Process Fuel	RED Value		This Study (RED Result)		
		Typical	Default	Average	Min	Max
Wheat ethanol	Not specified	32%	16%			
Wheat ethanol	Natural gas CHP	53%	47%	23%	-10%	47%
Wheat ethanol	Straw CHP	69%	69%	37%	7%	59%
Straw ethanol	Not specified	87%	85%	45%	28%	62%

10.9. Summary

The RED and RTFO were both developed directly for assessing the sustainability and GHG balances of biofuel production, whereas the PAS2050 is the first methodology to provide an applicable and consistent approach to accounting for the GHG balance from any product and service. All three can be applied to bioethanol production from wheat grain and wheat straw, though using the same input data, each methodology provides a different result. This is due to assumptions and interpretations to how the calculations are carried out. The most important difference in the methodologies is how co-products are accounted for. This particularly concerns the decision to award substitution credits to co-products, or allocate upstream GHG emissions between them.

The different results are a consequence of differences in the calculation methodologies, due to the approach the methodology takes to LCA; whether the method tends toward attributional or consequential LCA (Brander et al. 2009a). For reporting purposes, the RED states that ALCA is best as it provides a snapshot of emissions that are released, and attributable to the production

and use of the product or service. CLCA, on the other hand is better suited for policy analysis as the potential impacts are applicable to a wider, even global scope. **None of the methodologies completely adheres to ACLA nor CLCA.**

If interpreted literally, the PAS2050 method is the less convoluted method, as it does not provide specific circumstances for calculating emissions from biofuels. The original RTFO provides the most careful definitions of co-products, by-products and wastes. The RTFO will soon be completely integrated into the RED, and the definitions must then follow those in the RED. When this happens biofuel producers will be obliged to use the RED when calculating their emissions. **Currently, however, the RED is too vague to be practical for GHG reporting.** Key improvements and justifications of methodological decisions are needed, along with precise definitions of co-products, by-products and wastes as different interpretations of these will affect the results. Also, transparent default figures should be provided, which are referenced from published work.

Calculations involving co-products were the cause of most of the variation in the results between methods. In each of the methodologies, two methods of treating co-products are recommended (system expansion and allocation); so there is always scope for differences in interpretation. Allocation methods between co-products could become more important in future bio-refineries where a range of products are produced.

The RED allocates zero emissions to "agricultural residues" and "residues from processing" but allocates co-products by energy content. A significant problem with attributing no emissions to "residues" and "residues from processing" is that it implies they are a waste. Straw-bioethanol producers will therefore not have to account for the sustainability of their straw source. There is evidence that indiscriminate removal can cause losses of SOC from soil, which has been demonstrated here to significantly compromise the GHG savings of bioethanol. This is true whether straw is used as a fuel source for the process, or the bioethanol feedstock itself.

It is not possible to say which of the available methodologies is currently best suited for biofuels, as they all require either clarification or adaptation for biofuel GHG reporting. The most recent and accurate data should be used to correctly assess the impacts of the product or service. There are limited ways that the methodologies can ensure this is done. The methodologies should be careful not to combine ALCA and CLCA approaches so that the allocation methods and calculations provide meaningful information.

The following chapter examines these results in combination with those from Chapter 9 and 10 and discusses the main findings of this work.

Chapter 11. Discussion

This Chapter investigates the GHG emissions from 1st and 2nd generation bioethanol production. The variation in the results caused due to 'real' variation, 'uncertainty' or 'methodology' are analysed here. The three case studies focus on the conversion of wheat grain, wheat straw and Miscanthus into bioethanol via a life cycle perspective (**Figure 11-1**). The methods in which these analyses are carried out are described in **Chapter 3,** where the following main research objectives of this study are presented:

1. Develop a representative case study dataset in which to base LCA studies
2. Assess the sensitivity to parameters in LCA's of biofuels and agricultural products.
3. Identify impacts of system boundaries and emission factors on GHG results
4. Examine existing LCA methodologies used in GHG reporting methods.
5. Test the impacts of LCA methodology on the results.
6. Combine results of the analyses of variability and uncertainty and LCA methodology.

Objective 1 is fulfilled across **Chapters 3 to 8**, being introduced in **Chapter 1**, and the research methodology in which they are investigated is laid out in **Chapter 3.** The case studies are described in **Chapter 4** with full details of the goal and scope and system boundaries of the study. The methods and sources of data collected for the analysis is detailed in **Chapter 5**. The inputs, outputs and land use impacts of cultivating the crops are presented in **Chapters 6 and 7**. The process whereby they are converted to bioethanol is described in **Chapter 8**.

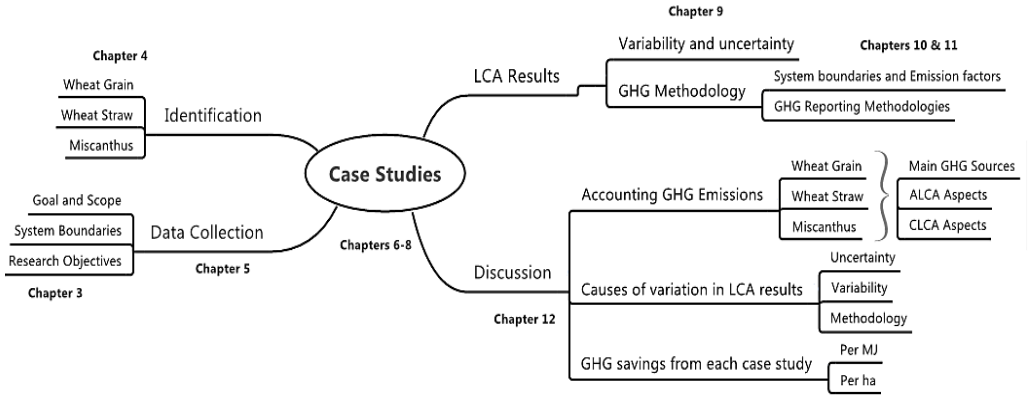

Figure 11-1. The relationship between the chapters in this thesis and the discussion topics of Chapter 12.

Objective 2 is addressed in **Chapter 9** where variation in crop GHG emissions is examined according to variability in farm inputs and outputs and uncertainty in GHG emissions from inputs and land use. This was tested to the farm-gate to avoid encompassing impacts of GHG accounting methodology on the result, as various co-products arise during the bioethanol production process.

Chapter 10 calculated the crop-scale GHG emissions according to some existing and publically available GHG assessment tools, addressing research objective 3. Objectives 4 and 5 are examined in **Chapter 11** where the full bioethanol supply chain was assessed according to, what would be considered to be a 'pure' ALCA approach for regulating GHG emissions from biofuels. The results from this were compared with the GHG reporting methodologies in existence. This chapter addresses the remaining 6th objective.

11.1. Accounting for GHG Emissions from Biofuels

In most accounts the average GHG emissions from 1st and 2nd generation bioethanol can currently fulfil the 35% emission saving targets set by the European RED. Some scenarios that may not reach stricter 50% and 60% targets include utilising natural gas-fired CHP units, straw if it leads to losses of SOC, and Miscanthus that is harvested in the autumn.

The average GHG emission from 1st generation bioethanol vary greatly between the RED, RTFO and PAS2050 GHG reporting methodologies: ranging from 22% when calculated by the RED with DDGS interpreted to be a residue, to 63% under the RTFO with DDGS awarded substitution credits for avoided animal feed (**Figure 11-2**). The higher emission saving result relies on using straw in the boiler, assuming it is removed from soil in a sustainable manner.

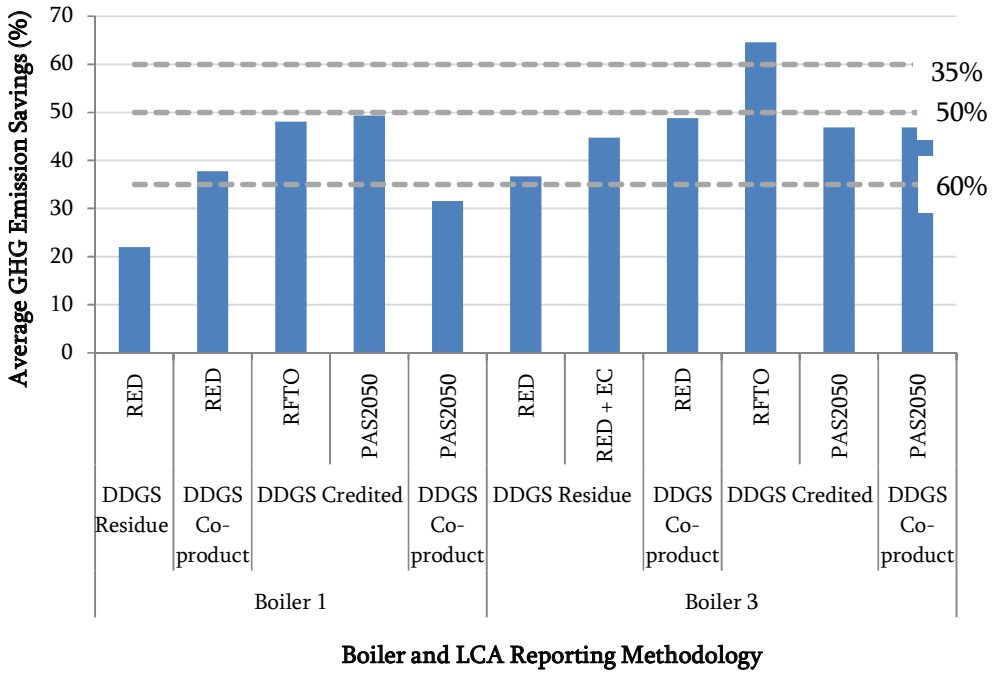

Figure 11-2. The average GHG emission savings from 1st generation bioethanol according to different GHG reporting methodologies.

The average savings from 2nd generation bioethanol range from 52 to 74% when calculated in a 'pure ALCA' and 53 to 91% when calculated according to the GHG reporting methodologies (**Figure 11-2**). The worse-case scenario is seen when using autumn-harvested Miscanthus. The best scenario utilises spring-harvested Miscanthus grown on arable land. The default numbers of the RED anticipate GHG emission savings from straw-based bioethanol to be 87%, which can only be achieved in this case study by applying the LCA methodology in the RTFO and PAS2050.

Differences between the GHG reporting methodologies are caused by the way that co-products are dealt with in the calculations. A pure ALCA would not tend to award substitution credits to co-products (Brander et al. 2009a), however this is performed in all of the methodologies examined. Due to uncertainty in determining the GHG credits for co-product substitution, the average GHG emission savings generated by the reporting methodologies show a larger range than the pure ALCA.

It is difficult to define the LCA calculation rules within the methodologies as some aspects are vague and open to interpretation. Different interpretations in the status of DDGS may affect the calculation methods and subsequently, the results. There is current confusion over the definitions of 'residues' from processing and this will be dealt with by using clearer language in a future updated version of the RED (ICCT, 2012).

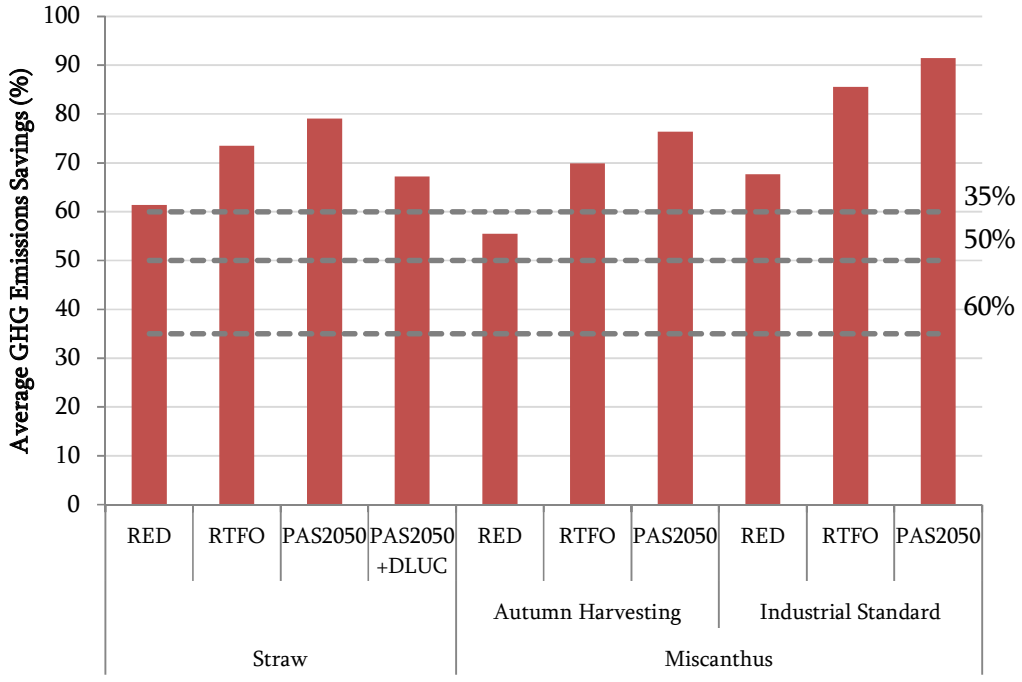

Figure 11-3. The average GHG emission savings from 2nd generation bioethanol according to different GHG reporting methodologies.

In both cases, the RED does not return the highest GHG emission savings. The RED aims to incentivise the use of straw by not allocating upstream GHG emissions to it. Despite this, the RED gave the lowest average savings (53%) because it does not award credits to exported electricity when it is produced from a 'residue from processing'. As a result, the GHG emission savings are 13% lower than awarding average electricity credits. Achieving higher GHG emission savings relies on GHG credits that are awarded to exported electricity from 2nd generation bioethanol, as they cannot be achieved without them. Also, the RED gives a lower than expect GHG emission saving in the 1st generation biofuel system when straw is used in the boiler. **Therefore, a major finding of this study is that the current RED methodology penalises the use of renewable energy and its calculation methodology does not support 2nd generation bioethanol production.**

11.1.1. Methodological Issues in GHG Accounting

This study has demonstrated that when the GHG emissions of the same process are measured by different methodologies, very different results can be seen. In **Chapter 11**, a 'pure' case ALCA was suggested based on the rules specified in the ISO Standards, considering recent opinions on what methodological options are appropriate for an attributional study. Despite this, the methodology followed in the pure ALCA differed considerably to those in the GHG reporting methodologies and therefore the results vary (**Figure 11-4**).

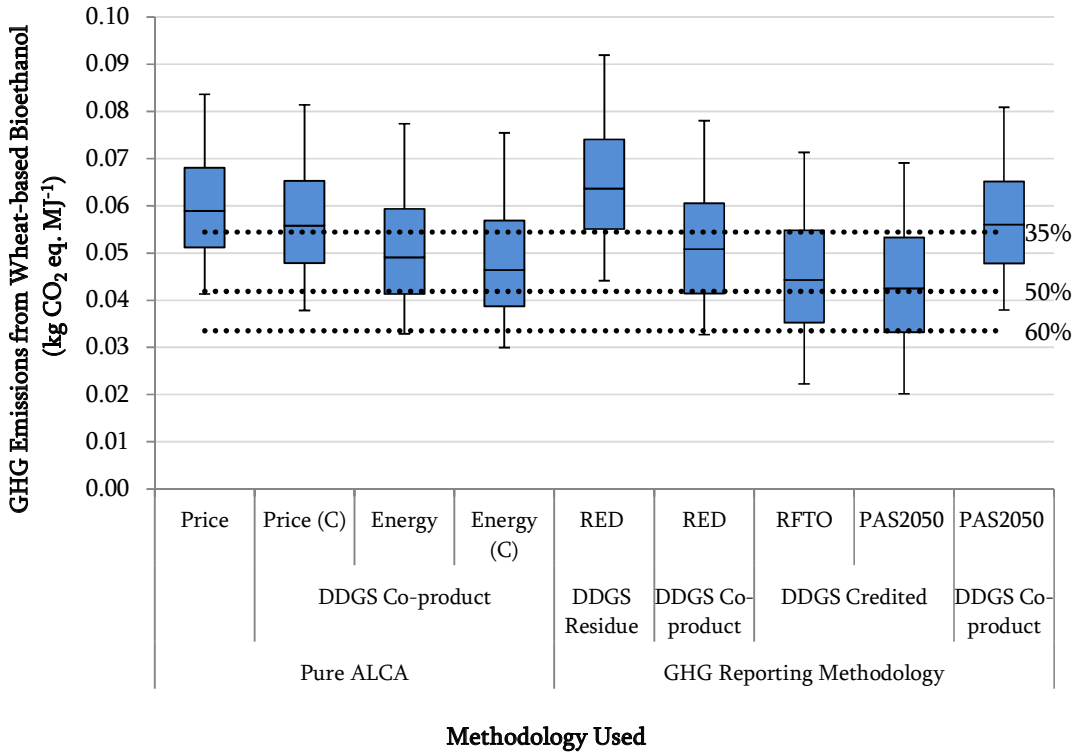

Figure 11-4 showing an example of outputs from a pure ALCA and from the GHG reporting methodologies for 1st generation bioethanol.

The primary aim of these methodologies was to provide a consistent approach to GHG reporting. **Another major finding of this study is that there is evidence that the GHG methodologies have deviated away from a 'pure ALCA' as the variation in the results is greater than that would be expected if this were followed.** The methodologies applied in policy have confused the appropriate use of ALCA and CLCA techniques.

From the results of this study, there appears to be three main ways in which methodologies could be harmonised: through agreement on the system boundaries of crops, harmonisation of emission factors and consistent means of dealing with co-products. The relevance of each of these points is discussed in the following sub-sections.

11.1.1.1. System Boundaries

The GHG emissions from 1st generation bioethanol are dominated by those that arise during wheat cultivation, whereas with 2nd generation biofuels, the major source of GHG emissions are from chemicals and enzyme requirements for pre-treatment, saccharification and fermentation (**Figure 11-5**). Other parameters show less importance in terms of overall sensitivity in the results. These parameters include pesticides, seed or rhizomes and machinery manufacture. These could be included or excluded from studies. Transport contributed on average 2-3% of GHG emissions. A nominal distance of 50km was assumed but the GHG emission results were not sensitive to transport.

This study offers a novel study of an industrial rhizome cultivation phase. It has shown that, despite relying on heavy cultivation during establishment and termination phases, over the whole life cycle of the crop rhizome cultivation has a negligible contribution to the final results. This is mainly due to the high yields achieved by rhizome multiplication. Rhizome costs of establishment are still the most significant component of establishment costs (Calu, n.d.). The reason for this discrepancy may be that there may be high costs for contractor work, fuel costs, land rent and labour.

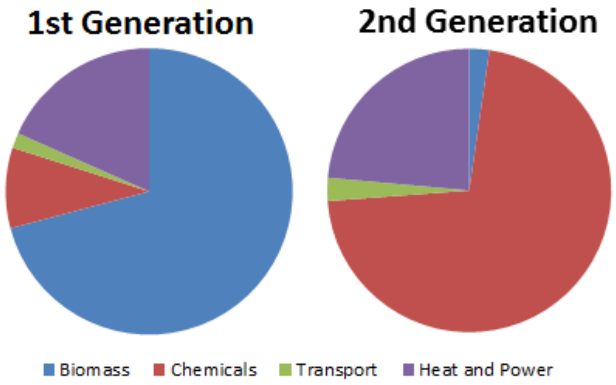

Figure 11-5 .The sources of GHG emissions from 1st and 2nd generation bioethanol.

Diesel fuel consumption represented an average of 7% of the arable crop system and literature-based Miscanthus study. In contrast this represents 52% of the 'industrial standard' Miscanthus as the impacts of fertilisers are omitted. Diesel fuel may therefore be an important aspect of accounting in biomass systems that do not require fertilisers. For example, diesel fuel requirements account for over 80% of the total GHG emissions from harvesting forestry residues (Whittaker et al., 2011). When interviewing farmers to complete the questionnaires, it was difficult to collect fuel data for operations because it either was highly variable, difficult to specify or carried out by contractors (M. Carver pers. com. 2012). **This study has highlighted that estimating diesel fuel consumption from theoretical equations showed considerable overestimations compared to literature (see Figure 6-3 of Chapter 6).** This is particularly true in operations that involve a lot of interaction in the soil, such as ploughing (Lal, 2004b). Therefore a suggestion would be to develop an acceptable range for these figures and provide them in an emission factor database.

Other studies find similar causes of GHG emissions. This study calculated an average GHG emission range of 37 to 67 g CO_2 eq. MJ^{-1}, or a saving of 28 to 56% compared to gasoline. Webbs et al. (2010) produced a regional assessment of wheat-based bioethanol production in the UK, to investigate how emissions varied with location. They found that emission savings ranged between 64%-71% due to differences in farm-based activities. Yan & Boies (2013) performed a similar study to that performed here, including temporal and spatial variability in wheat cultivation and uncertainty in emission factors from soils. They found that the final bioethanol emissions could range between 40 and 110 g CO_2 eq. MJ^{-1}, or a saving of 45% to an increase in emissions of 51%. The authors examine the full uncertainty range in the IPCC and conclude that a great deal of uncertainty is connected with N_2O emissions from soil and DLUC. Therefore it is generally understood that in arable-based LCA studies, N_2O and DLUC are major sources of GHG emissions. In this study the high level of variability of GHG emissions from cultivation means that the GHG emissions from 1st generation bioethanol range between -3% and 79% based on this factor alone.

Although few have studied the production of wheat straw to bioethanol, none have applied this to the current GHG reporting methodologies set out in the RTFO, RED or PAS2050. This study estimated that the GHG emissions from bioethanol from wheat straw ranged between 17 and 68 g CO_2 eq. MJ^{-1}, or a saving of 16 to 80%. The GHG emission savings are highly sensitive to methodological aspects regarding the credits awarded to exported electricity rather than the same cultivation-based aspects as seen in wheat grain. There are, however some serious uncertainties in the GHG implications with losses of SOC that have been shown to compromise the GHG savings of wheat-based bioethanol.

There are no specific studies that have investigated the GHG emission savings from Miscanthus to bioethanol supply chains. This study estimated that the GHG emissions from bioethanol from commercial Miscanthus ranged between 6 and 62 g CO_2 eq. MJ^{-1}, or a saving of 26 to 93%. Miscanthus is slightly more sensitive to K_2O and P_2O_5 and there are temporal sensitivities that are not seen in arable crops. The rotation period of the crop ultimately affects the amount of

biomass that the establishment and termination events are shared between, as harvesting is consistent on a per-tonne basis. This study assumes a rotation period of between five and 20 years. The observed uncertainty may have interesting implications for GHG accounting.

Examining the effect of Miscanthus rotation on the overall GHG emissions per tonne is novel as most LCA studies usually assume a fixed rotation period (e.g. Bullard & Metcalf 2001; Elsayed et al. 2003). There is no certainty that the fixed period will be fulfilled. A potential impact could be that if commercial farmer plans on having their crop for the full 20 year period, each tonne of Miscanthus will have an average emission of 13.5 kg CO_2 eq. If, for some unknown reason, they must abandon their crop after the minimal period then every tonne of Miscanthus that had been produced on that site would have at least twice the GHG emissions than they had expected.

The main question here is, if the Miscanthus had been sold and used as an energy crop then who would have to account for the resulting increase in GHG emissions? Would the consumer have to back-account for them? Considering that the results of the study suggest that the GHG emission savings from Miscanthus are not sensitive to rotation periods after five years, this may not be an issue for 35% GHG saving targets, but may be important when they become stricter.

11.1.1.2. Emission Factors

Emission factors are a crucial element of the impact assessment phase as they are used to convert the inventory data into a total emission to the environment. The analysis in **Chapter 11** showed that emission factors are important in determining the GHG emissions from the cultivation stage. A similar study by Hennecke et al. (2012), compared the Biograce tool (Biograce.net, 2012) with that developed by the Roundtable of Sustainable Biofuels (RSB, 2012). The authors found that differences between the tools were sufficient for the Biograce results to meet the 35% emission saving targets, but not in the RSB tool. The authors suggest that the differences are due to the emission factors adopted within the tools inbuilt calculations in the cultivation and processing stages. They suggests that this issue is a potential policy and methodological gap that should be addressed in future tools and methodologies (Hennecke et al., 2012). Transparency is beginning to emerge with the development of the BIOGRACE website (www.biograce.net) which provides more detailed information on emission factors that will be used to support RED calculations.

Calculation tools can help make complicated LCA calculations accessible to those with less expertise, but can also help harmonise calculations (Hennecke et al., 2012). This could be facilitated by developing a database of emission factors. For example, **Chapter 11** showed that differences in emission factors for nitrogen fertiliser caused variation in the results between calculation tools. This is particularly true for ammonium nitrate use, as it is much more 'GHG intensive' to manufacture compared to ammonium sulphate and urea (Brentrup and Palliere, 2008). This also presents a challenge to the data collection phase, as it is important to acquire more detailed data than simply 'N fertiliser use'.

It is also important to ensure that the methods in which the emission factors are calculated are also consistent. The emission factors themselves are derived from selective LCA studies for producing a given quantity of fuel or material to the factory gate or point of use. Therefore, the GHG emissions from these studies will also be susceptible to the methods in which they are calculated. This can be demonstrated by modifying the emission factor for ammonium nitrate fertiliser. The analysis performed by North Energy Associates (2006) examines ammonium nitrate manufacture using natural gas as a feedstock. They assume that is possible to recover 1.39 kg CO_2, with a value of £0.21 kg CO_2 per kg of ammonium nitrate (£8.44 kg^{-1}) from the process, which can be sold for industrial use. Price and mass allocation attributes 92% and 24% of the upstream GHG emissions to the end product, respectively, and the GHG emission factor for ammonium nitrate falls from 7.11 to 5.36 kg CO_2 eq. kg^{-1} as a result. This example cannot be placed in context with the full bioethanol supply chain as mass allocation is not performed; however it illustrates the importance that LCA methodology is applied to the emission factors also.

11.1.1.3. Co-Product Allocation

The ISO Standards claims that there are 'clearly stated procedures' for allocation (CEN, 2006b), but yet differences still are seen between studies. The ISO standards prefer that allocation is done via a physical relationship rather than by price (CEN, 2006b). Although 'mass' does describe a physical relationship between the two co-products, it is considered to be inappropriate as it is possible to improve the GHG emissions of a determining product by producing a large quantity of a low-value co-product (Reijnders, 2008a). This could be considered to be an ineffective way to incentivise the production of low-carbon products and is rarely carried out in LCA studies of bioethanol (Menichetti and Otto, 2008).

Energy allocation represents another physical attribute that is shared between co-products, though this may be considered arbitrary when both co-products are not used for the same end-use (Weidema, 2000). Energy allocation is 'overlooked' in LCA studies (Menichetti and Otto, 2008). It can reduce some uncertainty due to allocation as it does not fluctuate in the same way that price does. The results in **Chapter 11**, **Figure 10-5** indicate that energy allocation favours the GHG emission savings of 1st generation bioethanol. In contrast, allocation by price favours the production of 2nd generation biofuels (**Table 11-1**). Similar results are found in the literature (Gnansounou et al., 2009; Yan and Boies, 2013).

Table 11-1 showing average GHG savings from 1st and 2nd generation bioethanol with different methods of calculating for co-products.

Bioethanol Feedstock	Average GHG Savings (%)		
(Allocation Method)	Economic Value	Energy Content	Substitution Credits
Wheat Grain	44	50	81
Wheat Straw	62	57	72

If system expansion is used in ALCA then it should use average rather than marginal data (Sanchez et al., 2012). This may be a suitable option for co-products where there is an obvious displacement, (e.g. exported electricity obviously displaces grid electricity production). Average credits are awarded in the PAS2050 methodology but not in the RTFO or RED. The RTFO methodology applies marginal credits.

The RED applies credits based on the source of the electricity, which has been demonstrated to effectively penalise the use of renewable energy by not awarding the biofuel producer with a similar credit when they use renewable sources to lower the GHG emissions for processing. For example, if process heat and electricity is generated from wheat straw then an estimated 10 to 20 g CO_2 eq. could be avoided per MJ bioethanol. The GHG emission savings should be improved from 22% in the natural gas boiler to 45%, however because the same credits are not given to straw-generated electricity, the savings are only increased to 36%.

Issues of allocation also affect 2nd generation bioethanol production, mainly due to exported electricity. The GHG savings calculated under the RED accounting rules are 19% lower than if calculated according to the PAS2050 methodology (see **Figure 10-12** in **Chapter 11**). This again is because the RED does not award credits to exported electricity when generated from a 'residue from processing'. In contrast the PAS2050 rules award credits for avoided grid electricity to any exported electricity. There is a lack of reason to why this accounting rule exists in the RED.

The treatment of DDGS in the GHG reporting methodologies could be considered to be open to interpretation, apart from in the original RTFO, where it stated that system expansion is applied. The RED states that 'residues from processing' are not co-products, however there is evidence that DDGS is a valuable co-product from bioethanol, and some may account for it as a co-product. If accounted as a co-product the upstream GHG emissions are attributed by energy content and this changes the GHG emission savings from 47 to 60%. The PAS2050 methodology states that system expansion should be applied if possible; otherwise allocation by price should be performed: changing the GHG emission savings from 76 to 54%, respectively.

Determining substitution credits for DDGS is rife with uncertainty as it depends on the nutrient content and price of DDGS (Yan and Boies, 2013). This would also be an issue in biodiesel produced from oilseed rape, of which rape meal is a co-product from the conversion process (Elsayed et al., 2003). Likewise residual pulp from sugar beet-derived bioethanol can also be dried for use as an animal feed (Mortimer et al., 2004). Therefore 1st generation bioethanol appear to have an issue with accounting for this valuable co-product as deciding between co-product allocation or system expansion has a great effect on the estimated GHG savings.

11.1.2. Consequential Issues with 1st Generation Bioethanol

A CLCA measures the GHG emissions that occur due to a change in production of a product (Brander et al. 2009) and there are a number of 'consequential' elements present in the GHG reporting methodologies. This is further evidence that there is confusion between ALCA and CLCA within current policy applications which may be due to the way the debate has been framed in terms and direct and indirect effects (Brander et al. 2009a). The main consequential element is the use of substitution credits in ALCA, which has been discussed in the above sections. Other issues include the planned use of ILUC factors and the reason for incentivising non-food and non-land biomass resources. These are discussed here.

11.1.2.1. Introducing ILUC

Neither the GHG reporting methodologies nor the GHG calculation tools currently include ILUC in the system boundaries of their GHG calculations as it is difficult to calculate, predict and validate (EC, 2010a). There is some indication, however, that the RED intends on including this in further revisions (EC 2009a; ICCT 2012). It states that:

> *"The Commission should develop a concrete methodology to minimise greenhouse gas emissions caused by indirect land-use changes. To this end, the Commission should analyse, on the basis of best available scientific evidence, in particular, the inclusion of a factor for indirect land-use changes in the calculation of greenhouse gas emissions and the need to incentivise sustainable biofuels which minimise the impacts of land-use change and improve biofuel sustainability with respect to indirect land-use change."*

Modelling the GHG emissions that arise due to ILUC requires a coupled modelling framework that estimates interaction of agricultural and energy markets, predicts production volumes and prices, and estimation of risk of conversion of land (Bauen et al., 2010; Witcover et al., 2013). The GHG implications of ILUC will also depend on the types of land conversions occurring (Searchinger et al., 2008). Models can either be based on economic models (Hiederer et al., 2010), patterns of land use change (Bauen et al., 2010) or statistics (Kim and Dale, 2011).

There are various models and reports on predicting emission from ILUC, though the focus is on '1st generation' biofuels (Bauen et al., 2010; Dehue et al., 2009; EC, 2010a). It is highly possible that different methodologies will also develop varying ILUC factors (**Table 11-2**). For example there are ranges in the ILUC factors between the Renewable Fuel Standard in the US (EPA, 2010), the Californian Low Carbon Fuel Standard (CA-LCFS, CARB, 2009) and the European RED (Laborde, 2011; Marelli et al., 2011). They represent the ILUC factor in g CO_2 eq. MJ^{-1} biofuel made from specific agricultural feedstocks. As there are large uncertainty ranges and differences between regulations, it is difficult to provide an accurate ILUC factor with much degree of confidence (Witcover et al., 2013). From **Table 11-2** it can be seen that there are large ranges in the estimates for ILUC between studies.

Table 11-2. ILUC factors (g CO_2 eq. MJ^{-1}) bioethanol produced from various feedstocks.

Feedstock	Regulation			
	US-RFS2	CA-LCFS	RED (to be in future revisions)	
(Reference)	(EPA, 2010)	(CARB, 2009)	(Laborde, 2011)	(Marelli et al., 2011)
Corn	28 (18–42)	30 (18–44)	7 (4–9)	9–10
Sugarcane	5(4–12)	46 (32–57)	-	5–14
Sugar Beet	-	-	-	2–4
Wheat	-	-	-	12
Switchgrass	12 (7–20)	-	-	-

If these ILUC values were added to the wheat grain-based bioethanol, when calculated under the RED, the GHG emission savings would drop from an average of 22% to just 8%. Although Miscanthus is a 'non-food' resource that does not directly compete with food (Wiloso et al., 2012), it still occupies land and may create economic incentives for changes in land use, so it is not immune to issues of ILUC as (Cherubini, 2010a). Despite this, the RED still awards double credits to any lignocellulosic material. Applying the ILUC factors to Miscanthus (**Table 11-2**), the GHG emissions are reduced from an average of 47% to 24-39%; therefore this impact is not trivial. **If different methodologies exist then it will only increase uncertainty and variability of GHG emission saving targets.**

It has been suggested that wheat grain-based bioethanol actually has a lower 'risk' of stimulating ILUC (Bauen et al., 2010) as it produces DDGS as a co-product, which is an approved source of animal feed (Environment Agency, 2008). Therefore, in theory, one hectare of a biofuel crop will have a net effect of less than one hectare due to animal feed displacement (Bauen et al., 2010; RFA, 2008b). The extent of this will depend on assumptions made on which conventional animal feed is displaced, for example displacement of 1 tonne soymeal displaces 0.5 ha, whereas the same amount of feed wheat displaces 0.1 ha (Dehue et al., 2009). The DDGS co-product can, in effect, offset some of the pressure on agricultural land, but this is not apparently included in the ILUC factors destined to be used by the RED (Marelli et al., 2011).

It has been recommended that accounting for ILUC should not be exclusive to GHG reporting for biofuels, but be part of a wider, global framework that protects carbon rich and biodiverse lands from destruction (Nuffield Council on Bioethics, 2011). Considering that ILUC is more suited to CLCA, impacts of ILUC may need to be accounted for through some other mechanism. **As the RED is planning on including ILUC in future GHG regulation calculations it suggests it is confusing the role, or power, that regulation of biofuels has in this world-wide issue.**

11.1.2.2. Incentivising Non-Food Biomass Resources

Concerns over potentially devastating consequences on net GHG emission savings caused by ILUC has meant that the use of 'non-food' and 'non-land' resources has been encouraged in the RED. It states that:

"For the purposes of demonstrating compliance with national renewable energy obligations placed on operators and the target for the use of energy from renewable sources in all forms of transport referred to in Article 3(4), the contribution made by biofuels produced from wastes, residues, non-food cellulosic material, and lignocellulosic material shall be considered to be twice that made by other biofuels."

Therefore, the RED awards 'double credits' to producers of lignocellulosic-derived biofuels (EC, 2009a), and these are set to be quadrupled for wheat straw in future revisions of the RED (ICCT, 2012). These credits are based on targets once the RED is fully implemented, rather than credits in the same sense of 'substitution credits' in LCA methodology. For example, if a Member State produces 5% of renewable fuels from food crops and 1.25% from lignocellulosic wastes, then if the 1.25% contribution is quadrupled, then that country has met their 10% biofuel contribution target (ICCT, 2012). The results show that the GHG emission savings from 2nd generation biofuels are not as high as anticipated by the default numbers of the RED, therefore the policy may not achieve the expected GHG savings.

It is debatable whether the RED's approach to incentivising the use of residues is a correct one. Not allocating upstream GHG emissions to straw is considered to be "clearly incorrect" (Kindred et al., 2008a). It may be more appropriate to judge the GHG emissions from wheat straw based on allocation. In this case the GHG emissions from producing straw would be assessed fairly, rather than arbitrarily awarding them with a zero GHG cost. Allocating GHG emissions provides some information on the product system. In the case of allocation by price it will reflect the main incentives for wheat production, which is to produce grain. As a result, the GHG emissions are shared between grain and straw according to their relative value.

It appears that a consequential concern over ILUC has led to a potential attributional problem, in that if GHG emissions are not allocated to straw, it implies there are no sustainability impacts associated with using it. Allocating zero GHG emissions to wastes and crop residues puts them equal in terms of GHG attractiveness as feedstocks, which is misleading (Aylott et al., 2012). This somewhat a contradiction of the European 'Common Agricultural Policy' which identifies cereal residues as an important contributor towards erosion control due to rainfall and wind (Louwagie et al., 2009) as well as being implicated with nutrient recycling, maintaining soil structure and regulating water retention (Lal, 2008b). Indiscriminate removal can lead to a decline in soil quality, which can have both short and long-term adverse impacts on the environment (Lal, 2005). Currently it is not apparent whether the RED includes GHG implications from removing straw from land; however it has shown to be a significant source of GHG emissions that severely compromise the GHG emission savings of straw-based bioethanol (**Chapter 11**, **Figure 10-11**).

As straw is a limited resource, it is important that it is used efficiently and effectively (Kindred et al. 2008a). It is important that GHG reporting does not distort practices that may lead to a compromise in soil fertility and function (Bhogal et al., 2007). A certain reliance on farmer-expert knowledge is required for them to follow residue management methods to best comply

with the EU Common Agricultural Policy. It is also the case that straw retention can lead to problems with pests (HGCA, 2009), emergence of seedlings (Morris et al., 2009) and nitrogen immobilisation (Limon-Ortega et al., 2008), therefore GHG reporting should not also penalise against straw removal for avoidance of these issues.

The determination of residue retention limits must be identified for individual sites in order for soil amendment to be achieved alongside providing a renewable source of fuel without competing directly with food crops. It is suggested, however, that even small removal rates of 20-40% can cause losses in SOC, which have been demonstrated to significantly compromise the GHG saving potential of lignocellulosic biofuels (Reijnders, 2008b). A key issue is determining the likelihood that bad practice by farmers leads to soil damage, as this will indicate whether GHG reporting should include impacts of straw removal. There is evidence that current practices have caused losses in SOC of between 4% and 23% in the UK due to increased cultivation intensity (King et al. 2005), and there is currently predicted that agricultural production will become more intensive in the near future (Mondini and Sequi, 2008). Residue management may therefore become more important for the long-term sustainability of agriculture; therefore it should be regarded as an important issue in the sustainability of biofuels.

11.2. Causes of Variation in LCA Results

The problem statement in **Section 1.8** of **Chapter 1** stated that variation in LCA results caused by either variable or uncertain inputs and outputs can be classified as that caused by known or known unknown parameters. These parameters can be described by measurement and scientific knowledge and there are options to reduce variability by using representative averages or default numbers. Variation in LCA methodology, on the other hand causes uncertainty due to 'decisions' (Björklund, 2002), which can be considered to be 'arbitrary' (Ekvall and Finnveden, 2001). Indeed, from the analysis in **Chapter 11** it can be seen that there are some arbitrary rules in the GHG reporting methodologies that appear to affect the results greatly. There is a need to understand how important such decisions are on the final GHG emission result, and compare it to the relative importance of variation and uncertainty. This is examined here by combining the outputs from **Chapters 9** and **11**, which addresses research objective 5.

Chapter 8 showed how the GHG emissions from wheat and Miscanthus vary due to variation or uncertainty. This was examined by manipulating the input data to develop an additional series of Runs. These are listed in **Table 11-3** and the major sources of variation were DLUC, N_2O emissions from soils due to emissions and ranges in fertiliser application. Highlighting the cause of the variation can identified where variation in the results can be reduced by increasing knowledge of the system. This variation is referred to as 'scientific uncertainty' and is discussed in the following sub-section.

11.2.1. Reducing Scientific Uncertainty in LCA Studies

There is some scope for reducing the level of uncertainty seen in the results. For example, there was some uncertainty in the type of fertiliser used on the farms because the questionnaires provided 'tonnes N' rather than specifying the exact type of fertiliser they used.

11.2.1.1. Uncertainty in N_2O Emissions from Soils

Uncertainty in N_2O emissions from soils could be reduced by adopting a higher-tier approach to calculating N_2O emissions from soil. Guo et al. (2011) found the DNDC model estimated that between 0.28% and 0.39% of applied N was released as N_2O under the field conditions studied. Likewise, another study found a 1.7-fold overestimation by IPCC Tier 1 compared to the DNDC model (Brown et al. 2002). A similar study by Yan & Boies (2013) showed that applying DNDC estimates caused the mean result to decrease by 10-20%, with reductions of the overall level of uncertainty by 18-26%. Modelling with DNDC requires data on soil composition, such as clay and organic content, meteorological data such as annual rainfall, average temperatures, and detailed accounts of the fertilisers used and when they were applied (Brown et al. 2002). Likewise, adopting higher Tier IPCC emission factors requires knowledge of soil pH, texture and drainage (Bouwman, 1996). There is a therefore a trade-off between the increased certainty in the GHG emission result and data availability. Practically, much farm-based data collection will be performed by farmers and increasing data availability may require additional time. There would need to be a clear benefit to incentivise increased efforts of data collection. Performing this in practice may therefore be challenging. Another solution may be presented in 2014, when it is expected that a series of UK regional maps of local N_2O emission factors will be developed (Whitaker et al., 2010).

11.2.1.2. Fertiliser Requirements of Miscanthus

The modelled GHG emissions from Miscanthus are highly variable due to a combination of variation in inputs due to uncertainty in fertiliser requirements of the crop. According to the current cultivation methods in industry, commercial scale Miscanthus does not require N addition, and reports an average yield of 8 tonnes ha^{-1}. When trying to validate such data from literature evidence, a large range of estimated fertiliser requirements are found. The GHG emissions are highly sensitive to nitrogen application rates, so any yield response would need to be significant to compensate for them.

True nutrient requirements must be determined in order to reduce the uncertainty in the GHG emissions from Miscanthus. The industry standard Miscanthus had a GHG emission rate approximately one quarter of that of the estimates from literature therefore this impact is significant.

It has is suggested that organic fertilisers should be used in Miscanthus cultivation as it helps maintain both a low GHG balance and cost (Cadoux et al., 2012). In this study, the methods applied in order to examine organic fertiliser have produced some interesting results. It shows that artificial fertilisers give a beneficial GHG balance compared to using organic fertilisers. Here, K_2O is the limiting nutrient in typical organic fertilisers (Smith & Slater 2010), therefore

if application rates are modified to satisfy a given requirement then there will be an excess of N applied to the soil.

The basis of this analysis may be incorrect however, as if organic fertilisers are applied to Miscanthus, it not likely that a given nutrient rate will be applied, because calculating such requirements are difficult. Also, the quantities of organic material required to fulfil the hypothesised K_2O requirements of the plant would be significant. Therefore, it is suggested here that in practice, it is more likely that organic fertilisers will be applied at lower rates throughout the rotation (Felten et al., 2013), however there is little data available on this and the industry standard scenario does not include any such additions. As the results show a high sensitivity to N application then any addition will increase the net GHG emissions from Miscanthus.

11.2.1.3. Direct Land Use Change

The GHG implications from DLUC are also large and uncertain. Again, as with N_2O emissions from soils, the level of uncertainty could be reduced by performing more detailed assessments of SOC changes due to land conversion or changes in management. Such assessments could be performed using processed based models for estimating specific changes in SOC, such as Roth C (Hillier et al., 2009) or the C-Tool Model (Petersen et al., 2002).

The DLUC implications of changes in residue management may assume that a change in residue management has occurred. This may be a critical issue. There are various markets for straw and farmers will respond to market forces when deciding whether to bale or incorporate it. Straw removal may then differ from year to year, making it difficult to attribute changes in SOC to a single crop.

The results in **Chapter 9** showed that the DLUC estimates following methodology provided in the RED and IPCC produce a higher estimate for carbon sequestered under Miscanthus than the rate estimated after literature review in this study. The RED and IPCC do not include residual sequestered carbon under the crop after termination. Very few studies have been performed on stands that have reached commercial 'age'. A study by Duffosé et al. (2012) terminated a 20-year old stand of Miscanthus and estimated that 34% of the stored carbon was lost due to oxidation of SOC during the termination phase. **Expectations of Miscanthus sequestration in current policy may therefore be overestimated.**

11.2.2. Variability and Uncertainty vs. Methodology

A number of run scenarios are developed in order to examine the effect of variability and uncertainty with that of LCA methodology. This is performed specifically for wheat-based bioethanol, then the overall range of GHG emissions according to the biomass input, conversion data and LCA methodology are compared for 1st and 2nd generation biofuels. **Figure 11-6** shows the GHG emission results according to 7 scenario runs for wheat-based bioethanol where straw is used in the boiler.

In this figure, the interpretations of the methodologies are as follows:

- **RED** – straw is a residue, DDGS is a residue, exported electricity is awarded credits according to displaced straw-based electricity
- **RTFO** – straw is allocated by price, DDGS is awarded credits for displaced animal feed, exported electricity is awarded credits for displaced marginal electricity
- **PAS2050** – straw and DDGS are allocated by price, exported electricity is awarded credits for displaced average grid electricity

The Runs are detailed in **Table 11-3**. In Runs 1-6 the average IPCC default of 1% kg N-N_2O kg N^{-1} is assumed. All of these runs are presented according to the three GHG reporting methodologies.

Table 11-3. Description of runs contributing to the analysis of variance, uncertainty and methodology on LCA results.

Run	Run Examines	Description of Variance Tested
1	Spatial and temporal variation	Variation in site inputs, yields, from farmer data
2	Variation in conversion data	Variation in heat, power and chemical use
3	Uncertainty in emission factors	Fertiliser emission factors
4	Uncertainty in system boundaries	Including/excluding 'major' and 'minor' sources of emissions in the system boundaries
5	Uncertainty in DLUC (on arable)	SOC losses due to straw removal
6	Total plus soil N_2O emissions	Default IPCC
7	All plus modified N_2O emissions from soils	Reduced N_2O emission rates according to literature

For each scenario run it is clear that each GHG reporting methodology has generated different results. Overall, the range seen in the RTFO is greater, due to the use of substitution credits. The fact that 100% of the cultivation GHG emissions have been allocated to bioethanol in the RED means that the GHG emissions are overly higher. Run 1 shows that the effect of variability due to inputs and outputs, such as fertiliser usage, diesel fuel and yields causes more variation in the results than due to the methodologies alone.

Runs 2, 3 and 4 show that the uncertainty in conversion data, emission factors and system boundaries is modest compared to those seen in Runs 1, 6 and 7. Run 5 shows how the uncertainty of DLUC impacts of straw removal increases the overall results. Runs 6 and 7 show the full IPCC range and those based on the findings of Guo et al. (2011). These assume a fertiliser induced emission rate between 0.28% and 0.39% of applied N was released as N_2O. It can be seen that this has a large effect of reducing the variability observed in Run 7. The differences caused by methodologies become more marked. **Therefore this suggests that if variation in scientific uncertainty is reduced, methodology becomes the major cause of variation.**

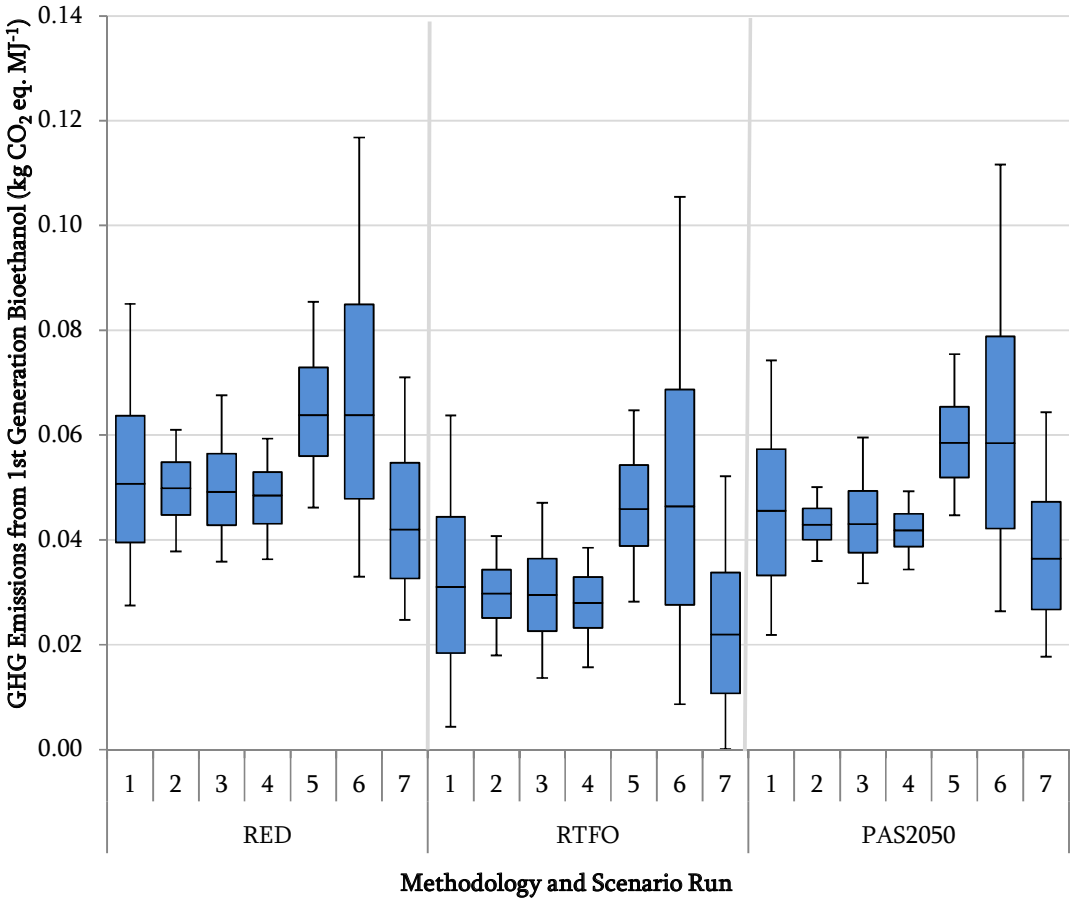

Figure 11-6. GHG Emissions from 1st generation bioethanol production from wheat grain under 7 scenario runs that examine different aspects of variability. Details of these runs are provided in Table 11-3.

Figure 11-7 shows the overall variation in the results from 1st and 2nd generation bioethanol caused by cultivation of the biomass feedstocks, from conversion to bioethanol and due to LCA methodology. These reiterate the results of Section 12.1, that in 1st generation bioethanol, the biomass feedstock appears to cause the majority of the variation in the result, which is due to N_2O emissions from soil with varying fertiliser applications. A lower range is seen in the 2nd generation biomass feedstock, even when including DLUC from changes in residue management and Miscanthus grown on arable land. There is less uncertainty in Miscanthus and wheat straw production compared to wheat grain.

For 2nd generation bioethanol production, the major cause of variation in the results is from conversion, due to both a high overall contribution from conversion and uncertainty in the bioethanol yield. In the 1st generation process most variation is caused by differences between fossil fuel and straw-powered generation.

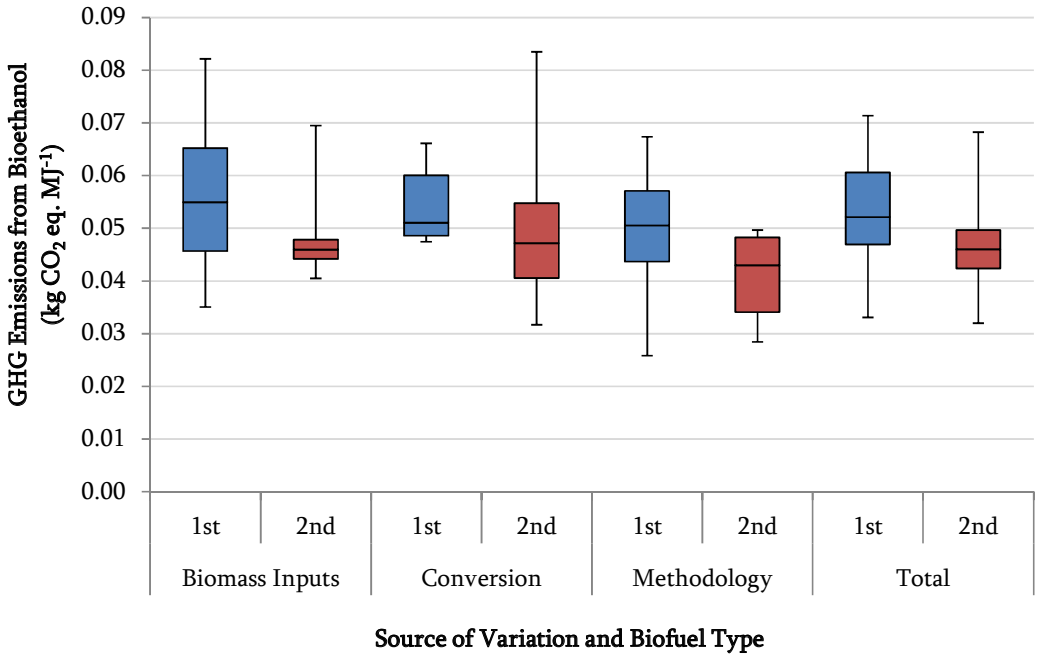

Figure 11-7. Causes of variation in 1st and 2nd generation bioethanol production.

The variation seen in the 'Methodology' category is comparable to the variation caused by biomass input in the 1st generation case study, and conversion in the 2nd generation case study. **This suggests that methodology is of equal importance and significance to scientific uncertainty when determining the GHG emissions from biofuels.** Therefore, one can assume that a great deal of variation between studies could be reduced if methodologies were harmonised.

Methodological variation is due to a combination of methods in which co-products are handled in the calculations. Previous results showed that there is a large uncertainty in the value of the substitution credits for animal feed (1st generation only) as well as differences in the GHG credits for exported electricity. Overall, the GHG emissions from bioethanol production range due to a combination of GHG emissions from biomass production, conversion and LCA methodology. The amount of variation is difficult to specify as there are sometimes overlaps between the categories.

Another comparison is made in **Figure 11-8**. It shows the relative spread of the GHG emission results for 1st and 2nd generation bioethanol when examining 'scientific variation' compared to that caused by LCA methodology. The main differences between the two are that in the 1st generation system, DDGS and exported electricity are co-products of bioethanol, whereas in 2nd generation it is only exported electricity.

It can be seen that the spread of the total results in the 1st generation case study is closely matched by the methodological variation, suggesting the GHG emission results are highly dependent on the methodology adopted.

In 2nd generation bioethanol LCA methodology increases the net variation of the results however the majority of the total results are affected by scientific uncertainty. This suggests that uncertainty in the GHG emission results for Miscanthus and wheat straw-based bioethanol influence the results more than methodological variation.

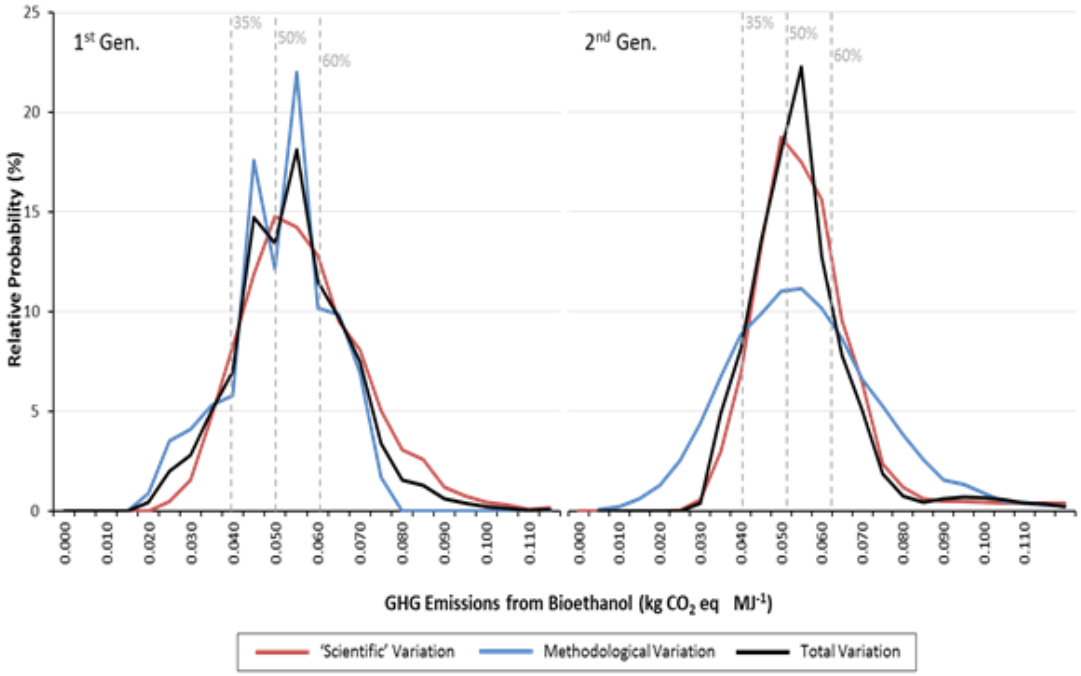

Figure 11-8. Comparing the effects of scientific and methodological variation on the total GHG emission results of 1st and 2nd generation bioethanol production.

11.3. Net GHG Savings from Biofuels

Three different 'types' of biomass resources were examined in this study and each has different issues that arise when accounting for the GHG emissions from their production and conversion to bioethanol. As a feedstock, the difference between wheat grain and wheat straw is that grain is a determining co-product, and straw is either a co-product or by-product of wheat grain production.

The supply of straw is inelastic as it is dependent on the demand for grain (RFA, 2010). Wheat is a food product and it's diversion to the bioethanol industry could place pressure on agricultural land which may stimulate land use change patterns that lead to a net loss of GHG emissions that can take many years to overcome, if at all (Royal Society, 2008; Searchinger et al., 2008). This impact may be somewhat lessened through the DDGS co-product which can be used as an animal feed (Bauen et al., 2010).

Figure 11-9 demonstrates the net outputs from one hectare of land if it is used to grow wheat for bioethanol production. One hectare of wheat yields between 1 and 4 tonnes bioethanol, equivalent to 27 to 110 GJ. From the area of land also 2.5 to 5.5 tonnes of straw can be acquired. Between 0.13 and 1.31 tonnes of straw can supply enough fuel to convert wheat grain to bioethanol, leaving between 0 and 3.6 tonnes on the site, or available for other uses, such as animal bedding. Between 1.3 and 4.8 tonnes of DDGS are also produced. DDGS is a co-product of bioethanol production. It can be sold and used as an animal feed, which is equivalent to 15-52 edible MJ ha^{-1}. Also, an export of 2 to 7 GJ of electricity is available from the conversion process.

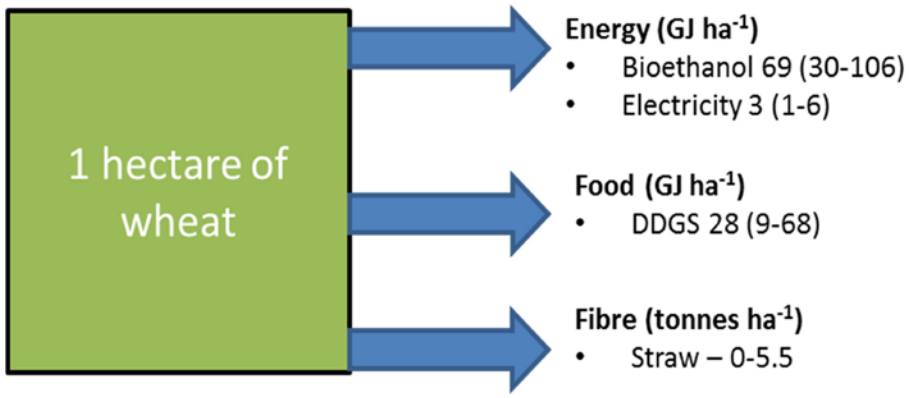

Figure 11-9. Summary of net outputs from 1 hectare of land used for 1st generation bioethanol production.

When the results are compared between each feedstock, very different outputs can be seen. These results are shown in **Table 11-4**, demonstrating the net outputs from one hectare of land if it is used to grow wheat or Miscanthus for bioethanol production. The highest source of each output of food, energy, fibre and the best GHG savings are highlighted. A calorific (edible) energy value of 10 MJ kg^{-1} and 15.7 MJ kg^{-1} for DDGS (ICCT, 2012) and wheat (Hughes, 2008) grain is assumed, respectively (also see **Figure 11-10**).

Table 11-4. Net food, energy and GHG savings from different bioethanol production options.

Land Use	Food (GJ ha^{-1})	Energy (GJ ha^{-1})		GHG Savings (tonnes CO_2 eq. ha^{-1})
		Bioethanol	Electricity	
Wheat Grain	28 (9 – 68)	69 (30 – 106)	3 (1 – 6)	34 (11 – 74)
Wheat Straw	**135 (72 – 208)**	24 (13 – 46)	1 (1 – 2)	10 (2 – 28)
Miscanthus (Autumn)	0	**86 (28 – 201)**	**7 (2 – 12)**	**45 (6 – 169)**
Miscanthus (Industry)	0	64 (20 – 137)	4 (8 – 2)	23 (1 – 89)

The results show that land can be used to provide different products, and in different quantities. There appears to be a trade-off between the potential for food and energy production from one hectare of land. Two of the scenarios could yield both food and fuel from land: 1st generation biofuel production from wheat grain yields DDGS as a co-product, and 2nd generation biofuel production from wheat straw yields wheat grain as a co-product. The latter scenario provides a higher net food output. When the primary target is to mitigate GHG emissions from the transport sector, the best option out of these case studies appears to be Miscanthus, particularly if it is harvested in autumn. This option provides the highest level of net energy (bioethanol and excess electricity) as well as providing the highest overall GHG savings. If the aim is to produce both food and fuel from the same area of land, the results suggest that 1st generation bioethanol can give the most beneficial ratio of food to energy and GHG emission savings.

There is some scepticism regarding the practice of autumn harvesting in Miscanthus. The sole benefit of doing so is that there is evidence that the yields are one third higher before the winter period. There is evidence that a higher nutrient off-take rate from the field occurs when harvesting in the autumn so the crop may become exhausted prematurely if this is carried out in succession (Clifton-Brown et al., 2007). This study explored how additional fertilisers could be added to the crop to compensate for harvesting losses but found that the GHG emissions from Miscanthus are sensitive to nitrogen application so harvesting in spring always gives better GHG emission savings. **Figure 11-10** however, shows that autumn harvesting may, in theory, provide higher net GHG savings because of higher output.

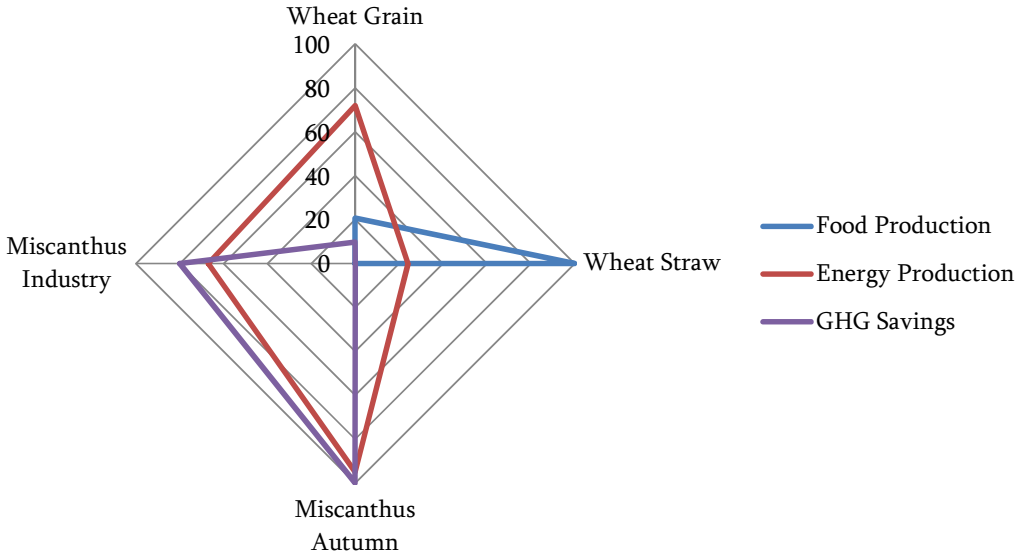

Figure 11-10. Net food, energy and GHG savings from different bioethanol production options.

There is general concern that 'green harvesting' of Miscanthus will lead to a decrease in rotation longevity (M. Carver pers. com. 2013). This study also showed that harvesting early would increase the GHG emissions per tonne of Miscanthus, and one would expected that it would increase the net costs of cropping, which are considered to be one of the limiting factors to uptake of the crops (Sherrington and Moran, 2010). Therefore, it appears that the best case scenario for a low-carbon crop is to harvest Miscanthus in the spring. It must be noted that other perennial grasses, such as *Arundo Donax* and Napier grass may not suffer from earlier harvests (M. Carver pers. com. 2013).

Applying the results of the ALCA to a given scale, say 1 hectare greatly affects the results, analysing the results in this way means that it becomes a different analysis altogether. By applying scale to the analysis, it changes the question from an attributional "what are the GHG emissions attributed to producing 1 MJ of bioethanol" to a consequential "what are the GHG emissions from producing bioethanol at a given scale". There are many more aspects, however, of a full CLCA that would need to be taken into account, such as the impacts that other co-products would have on existing markets, and indirect consequences of using land for bioethanol production. The different approaches therefore give different results. **This highlights the importance of the role of CLCA in renewable energy policy and ALCA in regulation of biofuels.**

11.4. Summary

Differences between ALCA and CLCA should be distinguished in regulation (Nuffield Council on Bioethics, 2011), by mixing approaches, it means that it is not clear how the result from such an analysis should be interpreted, being neither the attribution of absolute emissions nor the relative change in emissions resulting from a decisions (Brander et al. 2009a). The results show that there can be significant variability in the GHG emission savings when accounted according to different methodologies.

Each of the methodologies has different rules for how co-products should be accounted for in the calculations. The use of substitution credits instead of allocation has led to a large amount of variation in the results. This is because there can be a large degree of uncertainty when defining substitution credits; therefore this is not suitable for use in GHG regulation. For this an **attributional-based LCA** is best as it measures and regulates environmental impacts and attributes them to those that are responsible for their creation (Brander et al. 2009a; Sanchez et al. 2012). This concerns sources of GHG emission that biofuel producers have immediate control over (Nuffield Council on Bioethics, 2011). Indirect impacts are not considered as these tend to shift responsibility away from producers, who have less control on external events or the way the market responds to their activities. Such issues are considered to be relevant to CLCA. Therefore there is apparently some blending of techniques and specific rules in current methodologies that do not aid harmonisation between analyses.

It is evident from this study that the RED has aimed to incentivise the use of lignocellulosic material as a biofuel resource because it is anticipated that they will achieve large GHG emission savings of 85% and they do not directly compete with food. They incentivise the use of wheat straw by not attributing it with cultivation GHG emissions, but then straw is seen by producers as a 'CO_2 free feedstock'. There is evidence that if straw is used indiscriminately there may be adverse impacts on SOC which have been shown to compromise the GHG emission savings from 2nd generation biofuels. The RED also apparently penalises the use of renewable energy in processes by not treating exported electricity from renewable sources, equal to that produced from fossil fuels. Therefore despite the process benefiting from reduced energy requirements, the full benefit is not realised due to this specific rule, despite both cases electricity is exported to the grid and the source of the electricity should not affect the results. The scenarios in this study showed that higher GHG emission savings from 2nd generation biofuels could only be achieved when avoided electricity credits were awarded; therefore the RED is not fully supporting the use of 2nd generation resources.

Variation in LCA results caused by variable or uncertain inputs and outputs can be classified as 'scientific uncertainty' as it can be described by measurement and knowledge. Variation in LCA methodology, on the other hand causes uncertainty due to, sometimes arbitrary, decisions and the results have indicated that both of these forms of uncertainty have a large impact on the results. The amount of variation is difficult to specify as there are sometimes overlaps between the categories, however the results suggest that variation caused by methodology is comparable to the variation caused by scientific variation. The results also indicate that methodological variation becomes more important as knowledge of systems improve through increased measurements or detailed modelling. Options for reducing uncertainty due to methodology simply require harmonisation across methodologies and clear definitions.

Chapter 12. Conclusions, Recommendations and Suggestions for Future Work

The main conclusion of this work is that different LCA methodologies can cause significant differences in LCA results. This variation is particularly important to understand as it is largely caused by, sometimes arbitrary, decisions in how GHG emissions from a process should be accounted for and attributed to the main product. The variation caused by methodology is comparable to that caused by scientific variation, which is 'real' variation that sometimes cannot be avoided without detailed study.

It is important to understand how methodological variation causes differences in LCA studies so that harmonisation efforts can be focussed. This will help to increase the confidence in the GHG mitigation potential of biofuels. Unfortunately, this has not been achieved by the GHG reporting methodologies which are currently used for regulation in the UK. These include the European Commission's Renewable Energy Directive (RED), which is specifically designed to regulate the GHG emissions from biofuels, and the British Standards Institute's Publicly Available Standard (PAS2050), which is used for assessing products and services. The Renewable Transport Fuel Obligation (RTFO) is an artefact of previous biofuel policies in the UK and is hence becoming obsolete under the introduction of the RED.

The different results are a consequence of differences in the calculation methodologies, due to the approach the methodology takes to LCA; whether the method tends toward attributional or consequential LCA (Brander et al., 2009b). Neither of the methodologies completely adheres to ACLA nor CLCA. By confusing these two approaches they have failed to accurately attribute responsibility of GHG emissions from biofuel productions; therefore they do not fulfil their goal and scope.

In a 'pure ALCA', co-products would be identified as any product that has a monetary value, and they would be allocated emissions according to their energy content or price. The results show that differences between the two methods cause a small amount of variation; however it is the use of substitution credits in the GHG reporting methodologies that cause the majority of variation.

There are inconsistencies throughout the methodologies, and sometimes within them. For example, all of the methodologies treat co-products differently. In some cases they are allocated however substitution credits are recommended by the PAS2050 and RTFO methods. The RED arbitrarily assigns rules for specific co-products, for example agricultural residues are not attributed GHG emissions, nor are residues from processing, and this has been described as being "clearly incorrect" (Kindred et al. 2008a). It can be said that the current LCA methodology set within the RED is neither ISO-standard compliant nor fit for regulatory purposes.

Another main conclusion of this study is that the RED has failed to fully incentivise the development of 2nd generation biofuels from lignocellulosic feedstocks. Not allocating upstream GHG emissions to straw means biofuel producers will consider it a 'zero carbon' feedstock with no sustainability implications from sourcing it. The results of this study have shown that losses of soil organic carbon (SOC) due to indiscriminate straw removal could severely compromise the GHG emission savings from straw-based bioethanol. Also the GHG emission savings of 2nd generation biofuels are greatly improved if credits are awarded to excess electricity that is exported to the grid. Under the RED these are not awarded, or reduced credits are awarded if the electricity was generated from a renewable source. The reason for this is not clear, though biofuels are penalised if they use renewable sources of heat and power for onsite conversion.

This study has shown that the GHG emissions from Miscanthus are likely to be overestimated if they are based on data from literature. Commercial growers claim that the crop has minimal nutrient requirements, whereas in the literature there are a wide range of estimates. The overall GHG emissions from the crop are highly sensitive to nitrogen application; therefore any increase in use of N is disadvantageous. This study has also shown that early harvesting of Miscanthus to deliver greater yields does not improve the GHG balance if the nutrient requirements of the crop increase. There is also evidence that carbon sequestration rates under Miscanthus that are assumed in current policy are overestimated because they do not consider the residual amount of carbon after the crop has been terminated.

The results showed that in most accounts the average GHG emissions from 1st and 2nd bioethanol can currently fulfil the 35% emission saving targets set by the European RED. Some scenarios that may not reach stricter 50% and 60% targets include utilising natural gas-fired CHP units, straw if it leads to losses of SOC, and Miscanthus that is harvested in the autumn. The best 1st generation crop is produced using straw in the boiler, and the highest 2nd generation GHG emission savings were achieved by industrially produced Miscanthus. The results also suggest that allocation by energy content favours 1st generation biofuels, whereas allocation by price favours 2nd generation biofuels.

A LCA of a biofuel requires careful planning so that the model is fit to fulfil its original goal and scope. A comprehensive account will include emissions from soil and fertiliser manufacture in the system boundaries, as these represent approximately 80% of total emissions. Tools excluding these from the system boundaries will not provide a full account of the emissions resulting from arable cropping, therefore they should be avoided. Emissions from DLUC are another important source of GHG emissions that should be included in any GHG calculator that utilises land. Emissions from the use of diesel can be 'drowned out' by those from fertilisers, however in low-input crops this contribution can be large, for example it represents 52% of commercial Miscanthus. There is some uncertainty in this, however, as there are large discrepancies between theoretical diesel fuel consumption rates and those estimated in the literature.

12.1. Recommendations

The following recommendations are made for GHG reporting methodologies and the use of GHG calculation tools.

12.1.1. Recommendations for the GHG Reporting Methodologies

The following recommendations are made for how LCA should be used in GHG reporting methodologies:

- The LCA methodologies presented in the RED, PAS2050 and RTFO are not fit for the purpose of regulating GHG emissions from biofuels. They combine ALCA and CLCA methods, which must be addressed by re-writing the methodologies and all consequential aspects removed. A suggested rule should be that all co-products should be allocated and not awarded substitution credits.
- In LCA methodologies, particularly the RED, definitions of co-products, by-products and wastes should be clarified. This has the potential to cause differences in interpretations and there is a risk that a biofuel producer will select a definition that best suits their GHG calculation results.
- The RTFO uses a financial assessment to determine whether co-products are 'co-products' or 'by-products'. The RTFO then allocates 'co-products' by price if substitution credits are not appropriate. A better solution may be to allocate all co-products by price. If they are have a small value only a small proportion of upstream GHG emissions will be allocated to them. This would be a much fairer method to assess biofuel supply chains as it treats all co-products as equal.
- The LCA methodology applied in the RED does not conform to the ISO standards. It is highly arbitrary in the way that it treats crop residues and residues from processing differently to other co-products. This must be improved before the RED is implemented more legally-bound GHG regulation.
- The method in which the RED applies substitution credits to exported electricity is not correct and appears to actually penalise the use of renewable energy in the conversion process. Electricity that is exported to the national grid should be treated as equal despite its origin.
- The ISO standards are used to verify LCA studies and methodologies, yet they are too vague to ensure any harmonisation is achieved across studies. They do not provide sufficient guidelines in which to perform a LCA as they do not differentiate between ALCA and CLCA approaches. The ISO standards alone cannot ensure that LCA is carried out correctly. They must either be updated or an additional standard must be written to specify the key methodological differences between these two LCA approaches.
- The use of ILUC factors should not be introduced into future revisions of the RED. They will not only further confuse CLCA and ALCA aspects in biofuel regulation, but will increase uncertainty between studies that do/do not include them. If different resources are used to calculate ILUC factors then differences can arise here too. Attributing biofuels with

ILUC factors will also not recognise that land use is a wide and global issue that must not be attributed solely to biofuels.

- The impacts of straw removal can severely compromise the GHG emissions savings from straw-based bioethanol, or even where straw is used as a fuel source for heat and power generation. Straw should not be regarded as a 'waste' product as it is evident that it has key roles in the soil, including carbon incorporation and nutrient recycling.

12.1.2. Recommendations on GHG Calculation Tools

The following recommendations are made for the development of GHG calculation tools.

- Nitrogen fertiliser and N_2O emissions from soil must be present in the system boundaries of GHG calculation tools. Tools that do not include these should not be used for GHG regulation or even for 'in house' use by farmers. By excluding these sources of GHG emissions almost 80% of the total crop GHG emissions are excluded. This does not provide a comprehensive account of GHG emissions from an agricultural crop.
- An updated and transparent database of diesel fuel consumption rates for farm operations should be developed for use in tools. In particular, operations involving interaction with the soil should be listed in the database, including range data where possible. The database should provide a more accurate account of diesel fuel consumption in agricultural crops.
- Tools should record some aspect of uncertainty in their results. This will be both more accurate and more useful to analysing the results of the GHG assessment of a crop.
- All tools should include impacts of direct land use change, as the GHG emissions from a hectare of land are highly sensitive to this. Including the impacts of straw removal on GHG emissions may not be appropriate here, however, as this will be highly site-specific and it is not right to penalise against removing straw when it is necessary to control pest or disease problems. Care should be taken by farmers to ensure that straw removal rates are sustainable.
- Tools should use intelligent N_2O emission factors where possible. Although this may require more data entry to the tool it can reduce uncertainty in the results by one third. When UK-specific N_2O emission factors are available these should be implemented into all tools.
- Tools should be transparent and clearly state the sources of GHG emissions included in their system boundaries.

12.2. Suggestions for Further Work

Suggestions for further work would include the following:

- Research is needed to fully understand the implications of straw removal on soil and understanding the likelihood that financial incentives for 2nd generation resources will lead to non-sustainable removal rates. This would identify the soil types in the UK and Europe that are particularly sensitive to losses of SOC from straw removal, and identify improved management and cultivation methods that can compensate for losses of carbon in straw.

- Development of an up-to-date emission factor database is recommended. This would include all major used energy, chemical and materials used in agricultural systems, transport and biorefineries. This could expand to include processes involved in other bioenergy applications (heat or power) and could further expand to LCAs performed in other sectors. The database should be available to any LCA practitioner, be peer reviewed and include a transparent account of the methods in which the emission factors were developed.
- Some more research must be performed on the implications of green harvesting Miscanthus. The analysis in **Chapter 11, Section 12.3** showed that despite it having a lower GHG saving potential than spring-harvested Miscanthus, the sheer increase in biomass meant that overall a benefit of harvesting in autumn could be seen. The effect of early harvesting on the longevity of the crop is currently highly uncertain. There could be considerable calculation and financial implications if Miscanthus crops do not survive for at least 5 years.
- A suggestion would be to examine other energy grasses, such as *Arundo Donax* (giant cane) and *Pennisetum purpureum* (Napier grass) that could be harvested green without affecting the long-term longevity of the crop.

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Chapter 13. Appendices

Appendix 1a – Published Paper:

Appendix 1b – Published Paper:

Appendix 2 – Emission factors used in this study

Appendix 3 – Parameters for calculating N_2O emissions from soils

Appendix 4 – Diesel fuel consumption for Miscanthus

Appendix 5 – Estimates for cellulose manufacture from two LCA studies

Appendix 6 –Methodology for estimating the probability density function.

13.2. Appendix 2: Published Paper

A Comparison of Carbon Accounting Tools for Bioenergy and for Whole Farms. Whittaker, C., McManus, M. & Smith, P. *Environmental Software and Modelling,* 2013: 46 pp. 228-239.

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A comparison of carbon accounting tools for arable crops in the United Kingdom

Carly Whittaker [a,*], Marcelle C. McManus [a,b], Pete Smith [c]

[a] Department of Mechanical Engineering, University of Bath, Claverton Down, Bath BA2 7AY, UK
[b] Institute of Sustainable Energy and the Environment, University of Bath, Claverton Down, Bath BA2 7AY, UK
[c] Institute of Biological and Environmental Sciences, University of Aberdeen, Cruickshank Building, St. Machar Drive, Aberdeen AB24 3UU, UK

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ABSTRACT

In light of concerns over climate change and the need for national inventories for greenhouse gas reporting, there has been a recent increase in interest in the 'carbon foot printing' of products. A number of LCA-based carbon reporting tools have been developed in both the agricultural and renewable energy sectors, both of which follow calculation methodologies to account for GHG emissions from arable cropping. A review was performed to compare 11 existing greenhouse gas (GHG) accounting tools produced in order to calculate emissions from arable crops, either for food or bioenergy production in the UK, and a multi-criteria-analysis was performed to test their relative strengths and weaknesses. Tools designed for farm-based accounting achieved a higher 'user-friendliness' score, however bioenergy-based tools performed better in the overall level of information provided in the results, transparency and the comprehensiveness of emission sources included in the calculations. A model dataset for UK feed wheat was used to test the GHG emissions calculated by each tool. The results showed large differences, mainly due to how greenhouse gas emissions from fertiliser manufacture and application are accounted for. Overall, the Cool Farm Tool (Hillier et al., 2011) was identified as the highest ranking tool that is currently available in the public domain. The differences in the results between the tools appear to be due to the goal and scope, the system boundaries and underlying emission factor data.

1. Introduction

1.1. Climate change and everyday products

Concern over world-wide climate change has led to an increased interest in identifying major sources and sinks of carbon and greenhouse gases (GHG). The UK is committed to providing annual GHG reports to the UNFCCC and European Union as part of its legally binding Climate Change Act, committing it to reduce total National GHG emissions by 80% by 2050, using 1990 emissions as a baseline (HM Government, 2007). National-level reporting involves assessing sources and sinks of emissions from various sectors including the energy, transportation, agriculture, forestry sectors and identifying land use changes that have occurred over time. Methodologies for GHG accounting on a National level have been developed through a widespread scientific panel of experts in the UNFCC and IPCC, and in parallel to this, there has also been a number of GHG tools developed to assess agricultural and forestry practices (Colomb et al., 2012) and changes in soil carbon due to land use change (Coleman and Jenkinson, 2008; Miao et al., 2011; Palosuo et al., 2012).

The Climate Change Act has 'galvanised' interest in sustainability issues over all sectors of the economy (Gadema and Oglethorpe, 2011). In 2008, the Department for Transport introduced the Renewable Transport Fuel Obligation (RTFO), which posed a "legal obligation on fossil fuel producers to produce or supply renewable transport fuel" and defines the GHG calculation methodology that biofuel producers must use to report their GHG emissions (RFA, 2010). Since then the European Commission's Renewable Energy Directive (RED, EC, 2009), has been developed by the European Parliament and the Council of the European Union as part of the Climate Change Package agreed in December 2008. The RED was established to promote the uptake of energy from renewable resources, and it provides targets for participating Member States to commit to (Whittaker et al., 2011). The RED introduces sustainability criteria that specify that areas of high

* Corresponding author. Sustainable Energy Research Team, Department of Mechanical Engineering, University of Bath, Claverton Down, Bath BA2 7AY, UK. Tel.: +44 (0)1225 385 164.
E-mail address: C.Whittaker@bath.ac.uk (C. Whittaker).

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carbon and biodiversity must be preserved and requires that any changes in land use due to biofuel production are accounted for. The RED states that the GHG savings from biofuels should be at least 35% before January 2017, 50% after, and 60% after January 2018 for installations that start on or after 1 January 2017. The RED specifies a GHG reporting methodology by which calculations must be performed, however there are some ambiguous aspects of the methodology such as the definition of co-products and residues (Whittaker et al., 2011). In the calculations the RED does not specify which 'standard conversion values' or emission factors should be used when performing GHG calculations, which may lead to differences in results between tools (Hennecke et al., 2012).

In the business and commerce sectors, the progression to a low carbon economy will require changes in the way organisations deliver goods and services (Carbon Trust, 2008). In 2011, the British Standards Institution (BSI) updated the Publically Available Specification (PAS, (BSI, 2011)) 2050:2008 methodology (BSI, 2008), which presents a consistent approach to accounting for the GHG balance from any product or service (Sinden, 2009). Over the last decade there has been an increase in the number of companies that have voluntarily claimed to have committed to GHG reduction strategies following PAS2050. The main driver of this is believed to prepare businesses for future carbon markets where GHG emissions are traded globally (Hall et al., 2010). The process of examining the emissions from a product allows manufacturers to track hot spots and test the relative impacts that in-house decisions have on the GHG impacts of their business (Hall et al., 2010; Plassmann et al., 2010). There is also some evidence that consumers are gradually becoming more environmentally aware, however there remains some confusion to the relevance and impact of carbon labelling schemes (Gadema and Oglethorpe, 2011; Schmidt, 2009).

Of all economic sectors in the UK, agriculture contributes around 9% of GHG emissions annually (DECC, 2012), and is a significant component of the lifecycle emissions of many everyday food and other products. For example, Unilever analysed several products throughout their whole supply chain and found that agriculture is responsible for a significant contribution of the total lifecycle GHG emissions, for example Walkers crisps attribute 36% of the total life-cycle emissions for a bag of crisps to the growing of the potatoes (Pepsico, 2012). Agricultural processes also contribute significantly to GHG emissions from 1st generation biofuel supply chains. For example cultivation represents about 32% of total GHG emissions of bioethanol produced from wheat (Mortimer et al., 2004).

Although clearly a significant component of the life-cycle emissions, there are challenging aspects of quantifying GHG emissions from agriculture, forestry and other land uses (McKone et al., 2011). This is due to the dependence of emission on pedo-climatic and management details which are subject to temporal and spatial variations over various scales; leading to significant uncertainty in GHG emission assessments (McKone et al., 2011; Colomb et al., 2012). A clear need has been identified for access to farm relevant GHG calculators which are usable by farmers and land managers but robust and credible enough to be used in supply chain assessments.

1.2. Assessing the environmental impacts of products

The increased awareness of the environmental impacts from agricultural activities has prompted the development of methodologies that account for impacts in a holistic way. Life cycle assessment (LCA) is a technique which has dominated this area of environmental impact assessment as it systematically accounts for all the impacts that arise during the production, use and disposal of a product (Plassmann et al., 2010). The ISO standards 14040:2006 (CEN, 2006a) and ISO 14044:2006 (CEN, 2006b), describe the main phases of performing a LCA. These include first defining the goal and scope to outline the main aim of the study, such as the functional unit and the final unit of measurement (CEN, 2006b). The goal and scope should also provide some detail of the system boundaries of the study. A combination of data collection, iterative analysis, literature review and expert knowledge is required to in order to understand which sources of emissions need to be accounted for in the analysis, and likewise for the inventory phase, as the level and quality of data collected must be sufficient. Finally, the interpretation of the results must reflect back on the original goal and scope of the study, and should only be used according to the intended application of the study.

The 14040:2006 and ISO 14044:2006 state that: "There is no single method for conducting LCA. Organizations have the flexibility to implement LCA as established in this International Standard, in accordance with the intended application and the requirements of the organization" (CEN, 2006a). In LCA, methodologies are required to determine the way environmental impacts should be attributed to the final functional unit, including allocation rules that specify how emissions are split between co-products and how these are defined separately from wastes. Without a defined reporting methodology, there are various options to how a LCA could be performed, and it is possible that two studies that examine the same functional unit could both comply with the ISO standards and yet provide a different result. The ISO standards leaves a great deal of scope for interpretation and flexibility to the LCA practitioner (Aylott et al., 2011; Whittaker et al., 2011).

The Intergovernmental Panel on Climate Change (IPCC) provides some specific calculation methodologies for accounting for the GHG emissions for a range of activities that take place in the agricultural (De Klein et al., 2006), energy (Gomez et al., 2006) and transport sectors (Waldron et al., 2006). The IPCC guidelines classify quantification methods into three Tiers. Tier 1 emission factors are relatively straightforward and are derived to be applicable at global or national scale. They tend to 'average out' much of the climatic and geographic variation that nitrous oxide (N_2O) emissions from soil are sensitive to, therefore provide a high-level estimate (Hillier et al., 2011). Tier 2 methods increase the level of detail by employing "smart" emission factors that are specific to particular technologies or regions. Tier 3 methods incorporate increasingly more complicated or involved methods such as process-based models or direct measurement, for example DAYCENT (Del Grosso et al., 2001) or DNDC (Li et al., 2011). Using such tools requires a greater understanding of soil and plant systems compared to using the national default values (Hillier et al., 2011). Therefore, with the Tiered approach, the IPCC offers the LCA practitioner a series of approaches with varying levels of data requirements. This approach is useful where the ease of use has to be balanced against refinement.

1.3. GHG accounting in the agricultural sector

Over the last few years a number of GHG calculators have appeared in the public domain for calculating emissions from either single crops or for whole farms, and many more may have been commissioned by private companies for use "in-house" (N. Mortimer pers. com. 2012). The tools are computational models that contain in-built data that can be manipulated to some extent to provide an individualised account of the GHG emissions for a given crop and user. Other LCA-based tools, such as SimaPro and Gabi can be used, though these require a higher level of input data compared to the GHG calculators discussed here. These tools can be used to provide information on a range of other environmental impacts in addition to GHG emissions, which is necessary to perform a full

Table 1
List of GHG accounting tools included for further review.

Producer	Tool	Access	Brief description	Final unit of measurement	Website/(reference to tool/manual)
Farm-based tools					
Country Land and Business Association	CALM	Online	Farm-based GHG reporting	Whole farm	http://www.calm.cla.org.uk (CLA, 2008)
SEE 360 Ltd	C-PLANv0	Online	Farm-based carbon reporting	Whole farm	http://www2.cplan.org.uk (C-Plan, 2007)
Manchester University	CCalc	Download spread sheet	Carbon calculator of LCA emissions along supply chains	User defined	http://www.ccalc.org.uk (CCaLC, 2012)
Climate Friendly Food	Organic Farmer Carbon Calculator	Online	Farmer and growers carbon calculator	Whole farm	http://cffcarboncalculator.org.uk/carboncalc (CFF, 2009)
Cool Farm Institute	Cool Farm Tool v1.1	Download spread sheet	Greenhouse gas calculator for farming	Whole farm/1 ha of farm/1 tonne of crop	http://www.coolfarmtool.org (Hillier et al., 2011)
Muntons	Muntons barley calculator v4	Download spread sheet	Barley carbon calculator	1 tonne of crop	http://www.muntons.com/(Muntons.com, 2012)
Bioenergy-based tools					
Biograce	Biograce calculator v4b	Download spread sheet	Harmonisation of biofuel calculations	Hectare of crop/MJ bioethanol	http://www.biograce.net/(Biograce.net, 2012)
Renewable Fuel Agency	RFA – RTFO Carbon Calculator v1.0	Download programme	Biofuel carbon calculator	Hectare of crop/tonne/MJ bioethanol	http://www.dft.gov.uk/publications/carbon-calculator/(Westphal et al., 2011)
North Energy Associates and AEA Technology	Biomass Environmental Assessment Tool (BEATv2)	Download programme	Biofuel supply chain GHG assessment	Hectare of crop/1000 L bioethanol	http://www.biomassenergycentre.org.uk/ (AEA Technology & North Energy Associates, 2008)
Round Table of Sustainable Biofuels	RSB Tool	Online	Individual stages of supply chain	Hectare of crop/MJ bioethanol	http://buiprojekte.f2.htw-berlin.de:1339/welcome (RSB, 2011; RSB, 2012)
Home Grown Cereals Authority	HGCA Biofuel GHG Calculator	Online	Biofuel GHG calculator	Tonne/litre/GJ of bioethanol	http://www.hgca.com/content.output/2135/2135/Resources/Tools/Bioethanol%20Greenhouse%20Gas%20Calculator.mspx (Woods et al., 2005)

from growers guides and from literature. Fertiliser N is provided by a combination of ammonium nitrate (60%), urea (39.5%) and farm yard manure (0.5%, FYM) to test whether tools differentiate between different N sources. The relevance of the input data was not the main focus of study, rather the differences in the calculated emissions when the same input data are used in each tool. To eliminate methodological variation, the final emissions were based on a 'per hectare' basis, assuming that all straw is incorporated into the soil. Therefore, any differences in how emissions are allocated between wheat and grain were not observed.

3. Results and discussion

3.1. Desk-based review

The desk-based review identified 31 resources that can, in some way, be used to calculate the emissions from crop cultivation, or from a specific aspect of crop production (Appendix 1). Sources originated from commissioned work to environmental consultancy groups, from governmentally and non-governmentally funded organisations, programmes developed from scientific research in Universities and from certification schemes. Eleven of these resources were selected for further review (Table 1).

Of the 31 resources, 15 were excluded as they are based in the United States (6), Australia (5), Europe (2), New Zealand (1) and Canada (1). Seven of the resources were process-based modelling software that could be applied to the UK (Appendix 2); and were excluded as they are typically used to examine specific processes in the soil. CPlan has two developed two tools, however the more recent version (CPLANv2) is not free to use. The Carbon Trust Carbon Footprint Expert Tool® (Carbon Trust, 2012) was also excluded from further review as this is only available to consultancies for a fee.

In the 11 tools selected, 6 are designed in order to calculate the emissions on a farm-level and 5 examine cereal cultivation as part of the bioethanol production pathway, therefore they represent a combination of tools designed to raise awareness, and to report GHG emissions on a farm and product level. All of the tools require data input that would all be known by any farm owner or manager. There are large differences in the system boundaries of the tools, and these are listed in the supporting material in Appendix 1.

The main aim of the majority of the farm-based tools is to educate farmers as to where emissions occur on their farm (Muntons.com, 2012) and identify GHG mitigation options so that they are in a position to accept future GHG emission reduction challenges (C-Plan, 2007). This can be for marketing, economic or ethical reasons or to understand how they can adopt practices that lead to better quality soils (CFF, 2009). This is to some extent, a similar goal of the bioenergy-based tools; except they are focused on accurately measuring the GHG balance of various stages of the biofuel supply chain. The Biograce, RSB and RFA Tools are specifically produced to aid biofuel producers calculate their GHG balance according to the reporting methodology laid out in the RED. Both the RSB's own methodology and that detailed in the RED are examined here. The following sections discuss how the goal and scope of the tools affects the structure and function of the tools.

Table 2
Example of the MCA criteria and scoring.

Criteria	Score			
	3	2	1	0
User friendliness				
Is the tool readily available?	Yes – online or ready to download	Yes – but requires installation	Yes – but requires permission/password	Not available
Is support available?	Yes – a support telephone number	Yes – a support email address	Less obvious	None
Are manuals provided?	Yes – detailed with data collection guidance	Yes – detailed manual	Basic instructions	None

Table 3
Example for site input data for cultivation of wheat.

Input	Type	Amount	Units	Reference
Site establishment	Ploughing (diesel)	28.3	Litres/ha	(Williams et al., 2006)
	Power harrowing (diesel)	17.9	Litres/ha	
	Planting (diesel)	8.3	Litres/ha	
	Rolling (diesel)	2.2	Litres/ha	
	Spraying fertilisers (diesel)	5.4	Litres/ha	Assume performed 3 times (Nix, 2011), fuel consumption from Williams et al. (2006)
	Spreading manure (diesel)	9.7	Litres/ha	(Lal, 2004)
	Total diesel (per hectare)	71.8	Litres/ha	
Fertiliser	Urea (46.4% N)	123	kg/ha	Total N demand for feed wheat (Nix, 2011)
	Ammonium nitrate (34.5% N)	330	kg/ha	
	Cattle slurry (3% N)	633	kg/ha	
	(Total N)	190	kg N/ha	
	Phosphate	60	kg P/ha	(Nix, 2011)
	Potassium	43	kg K/ha	(Nix, 2011)
Pesticides	Herbicide	1.3	kg a.i/ha	(Garthwaite et al., 2010)
	Fungicide	1.4	kg a.i/ha	
	Insecticide	0.01	kg a.i/ha	
	Growth regulators	1.1	kg a.i/ha	
	Total pesticides	3.81	kg a.i/ha	
Seed	Wheat grain	175	kg/ha	(Nix, 2011)
Harvesting	Combine harvester (diesel)	20	Litres/ha	
Machinery used	Tractor (150 hp)	To power the following:		(Williams et al., 2006)
	Plough, 1 m	1.2	Hour/ha	
	Power harrow, 4 m (PTO powered)	0.8	Hour/ha	
	Seed drill, 6 m (PTO powered)	0.2	Hour/ha	
	Roll, 6 m	0.2	Hour/ha	
	Sprayer, 24 m (PTO powered)	0.1	Hour/ha	
	Spreader, 12 m (PTO powered)	0.2	Hour/ha	
	Combine harvester (200 hp)	0.4	Hour/ha	
	Total tractor hours	3.3	Hour/ha	
Soil data	Soil texture	Medium		Select middle-range types
	Soil organic matter	$1.72 < \text{SOM} \leq 5.16$		
	Soil moisture	Moist		
	Soil drainage	Good		
	Soil pH	$5.5 < \text{pH} \leq 7.3$		
	Soil type	Inceptisol		(USDA, 2005)
	Average rainfall	840 mm		1971–2000 average for England (Met Office, 2012)
Yield data				
Yield (wheat grain @13.5% m.c)		7.7	Tonnes/ha	(DEFRA et al., 2010)
Yield (straw-incorporated)		4	Tonnes/ha	(Nix, 2011)

3.2. Multi-criteria-analysis and the goal and scope

The MCA showed differences between the tools for each of the four categories assessed (Table 4). Across the four categories, the averages differed between farm and bioenergy-based tools (Table 4). Farm tools achieved a higher score for 'user friendliness' and the bioenergy tools were generally rated more informative, comprehensive and transparent.

In terms of user friendliness, the highest rating tools were the CFF and CCalc Tools (78%), followed by the Cool Farm Tool and the HGCA tool (72% (HGCA, 2011)). These tools were rated accordingly due to ease of access, intuitiveness, flexibility of input units and support and guidance for using the tools. Poorer performing tools were lacking instructions, required a password or installation and lacked flexible or non SI units. In the 'informative' category the majority of tools were high scoring. The highest rating tool was the Cool Farm Tool (100%), mainly because it provides results in various formats and with a breakdown of all emission sources. Less informative tools did not provide a clear enough breakdown of emissions.

The highest rated tools in terms of transparency were the Cool Farm, BEAT and RSB Tools. The Cool Farm and BEAT Tools permit the user to access the original Excel-based calculations, including referenced sources of emission factors. The RSB Tool does not provide an Excel-based model, although the manual is highly transparent. Lower ranking tools did not provide sufficient details of the types of emission sources included in the analysis, or provide details on the sources of emission factor data. The most comprehensive tools assessed were the Cool Farm and RSB Tools, as these included LUC and adopted Tier 3 IPCC methodology. LUC was included in 6 out of 11 tools, and these were given a higher rating as LUC can potentially dominate GHG emissions in agricultural LCAs (Roches et al., 2010). The least comprehensive tools were those that did not specify whether they include N fertiliser manufacture and/or application, or did not state which N_2O sources were included. Only BEAT included some aspects of uncertainty.

The results of the MCA in some way reflect the main goal and scope of the tools. As it is more likely that farm-based tools will be used by non-LCA practitioners there is a level of user-friendliness expected (Colomb et al., 2012). Bioenergy-based tools are generally used by LCA practitioners, or those with expertise in the industry to assess whether a biofuel has reached its GHG saving target. These tools therefore demand a greater level of information, transparency and accuracy.

The main goal of some of the farm-based tools is to provide a calculation platform to educate farmers about GHG emissions occurring due to their activities and choices of management. Therefore the scope of these tools is the measurement of GHG emissions that occur on whole farm perspective and not necessarily a single crop. These tools require information on all site inputs and fuel use, as well as other farm-based activities such as woodland sequestration. Tools that work on a single crop level, such as the

Table 4
Results of the MCA for the 10 GHG reporting tools assessed.

Category	Farm based							Bioenergy based					
	C-Plan	CFF	Muntons	Cool farm	CALM	CCalc	**Average**	RSB	Biograce	HGCA	BEAT	RTFO	**Average**
User friendliness	67%	78%	56%	72%	67%	78%	**69%**	61%	56%	72%	39%	61%	**58%**
Informative	50%	83%	17%	100%	83%	50%	**64%**	83%	83%	67%	83%	67%	**77%**
Transparency	17%	50%	17%	83%	75%	50%	**49%**	83%	83%	42%	83%	42%	**67%**
Comprehensiveness	20%	10%	23%	90%	60%	33%	**39%**	90%	80%	37%	40%	43%	**58%**

Muntons and Cool Farm Tool, require information on all site inputs attributed to a single crop.

In bioenergy-based tools the goal and scope is to accurately assess the GHG savings compared to using conventional fossil fuels. For example, the Biograce Tool was developed in response to the introduction of the RED, which sets specific GHG reduction targets over a set of timescales that must be met. The higher demands for accurate reporting of GHG emissions from biofuel production may explain why these tools score higher on the informative, transparent and comprehensive categories in the MCA.

3.3. Comprehensiveness and the systems boundaries

Fig. 2 demonstrates the overall emission profile generated by each tool, based on the input data in Table 3. The emissions from LUC are examined separately. The results demonstrate variation in both the magnitude of the estimated GHG emissions per hectare and the emission sources included in the analysis. The only emission source consistent in all tools is diesel fuel consumption. The system boundaries are closely linked to the comprehensiveness of the tools, as this considers which major sources of emissions were included in the calculations: tools lacking major sources of emissions were scored low in comprehensiveness. It is important to note here that comprehensiveness does not necessary imply accuracy.

Four out of the six farm-based tools have a goal to 'accurately measure the GHG emissions from a farm'; however the 'comprehensiveness' score from the MCA suggests this goal is not achieved. Overall, on a per hectare basis, the total GHG emissions ranged between 606 and 3298 kg CO_2 eq. The average emission result calculated by all tools is 2239 kg CO_2 eq./ha, where the average for the farm tools (1642 kg CO_2 eq./ha) is almost half that of the bioenergy tools (2836 kg CO_2 eq./ha). This is mainly due to the lower results calculated by the C-Plan and CFF Tools. It is difficult to determine why these tools calculate lower GHG emissions, due to their lack of transparency. These tools also scored a relatively low score in comprehensiveness: the CFF Tool achieved the lowest score (10%) as it has apparently excluded N fertiliser manufacture and N_2O emissions from soil, which account for an average of 43% of emissions in the tools that include them. The C-Plan Tool was rated low (20%) due to both a lack of detail on which GHG emission sources are included, for example whether they include direct,

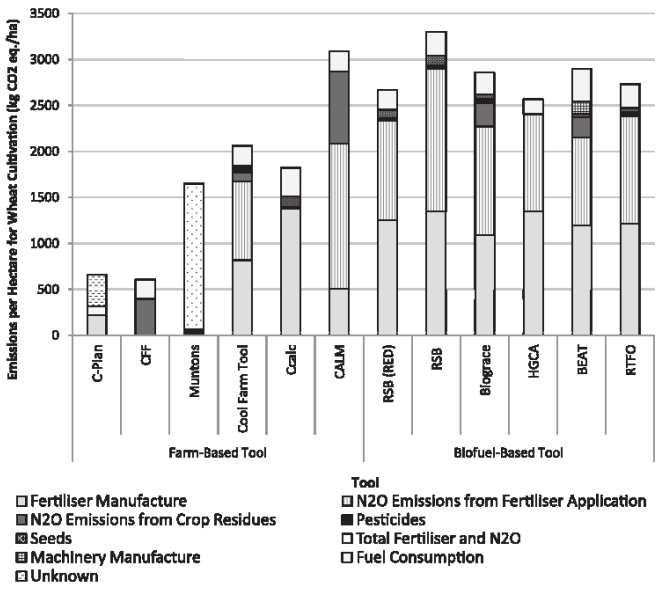

Fig. 2. Emission profile from each GHG accounting tool for 1 ha of wheat cultivation.

indirect N_2O emissions or those from crop residues, and also because their estimate for fertiliser production and application is lower than expected when using IPCC Tier 1 emission factors (Fig. 3). This tool has a separate entry for 'crops' in the results; however it is not clear what this specifies. Both the CCalc and Muntons tools have also been penalised in the comprehensiveness score due to the level of detail that they provide on N_2O emissions from soil.

There are various sources of emissions from soil (Fig. 3), whether these are from direct or indirect N_2O emissions from N fertiliser application, or N_2O from crop residue incorporation or manure application, or CO_2 emissions from lime or urea hydrolysis (De Klein et al., 2006). The variation in the emission estimates observed may be due to the IPCC Tier applied (Colomb et al., 2012), or due to the incompleteness of the N_2O calculations where there is a lack of knowledge of the various sources, or a combination of both. Fig. 3 indicates which tools calculate an emission result similar to that expected by the IPCC, suggesting that all sources of N_2O are included in their system boundaries. In many cases therefore, the issue of transparency and comprehensiveness could be improved if details of the system boundaries of the tools were stated on the tool developers' websites or manuals. Alternatively, tools should provide disaggregated results for emissions from soils.

Four tools did not include details on LUC (C-Plan, Muntons, HGCA, and BEAT). Those that do provide varying estimates for LUC, though this range is greater for LUC of forestland to arable land (Fig. 4). The order of magnitude of GHG emissions from wheat ranges between 606 and 3298 kg CO_2 eq./ha, whereas LUC ranges between 1918 and 7000 kg CO_2 eq./ha or 4147 and 27,000 kg CO_2 eq./ha for grassland or forestland conversion to arable land, respectively, therefore the range of potential impacts of LUC are highly uncertain, but overly large. The tools estimate different GHG implications of LUC, despite the Cool Farm Tool, Biograce and RSB Tools using the same original resource to calculate emissions (Bickel et al., 2006). This states that LUC carbon losses should be based on any land conversion that has taken place in the last 20 years and is dependent on the type of LUC occurring (Bickel et al., 2006). This methodology is also followed by the Commission Decision 2010/335/EU (EC, 2010), which users of the Biograce and RFA Tools are instructed to use. The Cool Farm Tool follows the IPCC to model specific LUC changes for over 113 countries (Hillier et al., 2011). The CALM Tool does not state the specific source of calculations; though it appears to produce a similar result to the Cool Farm Tool (Fig. 4).

For data entry of details of LUC, some tools require a selection of 'before' and 'after' land uses, whereas some require more details on the soil type, geographic zone and changes in how the site is managed. Out of the 7 tools that include LUC, the majority provide default drop-down menus as well as offering the user-defined inputs. Both the Cool Farm and RSB Tools provide a detailed calculation tool for LUC, including changes in tillage, inputs and residue management. The RSB Tool does this, although in a generally less user-friendly manner. Though it is outside the scope of this paper to determine the accuracy of LUC estimates, it is clear that the Cool Farm Tool offers both the most comprehensive and most accessible calculations for LUC for a non-LCA practitioner to use, though if the user can calculate their own LUC estimate then all tools that offer this function are appropriate.

Uncertainty was clearly lacking in the majority of the tools, as only BEAT included some indication on how the result may range. The CCalc Tool requires the user to define the level of data quality, but it does not appear to be referred to in the results. Also, none of the tools provide an account of which sources of GHG the final results are most sensitive to, but these are known from literature (Roches et al., 2010). An absence of uncertainty suggests a lack of comprehensiveness as this can provide some information on the robustness of the data sources used and detail any temporal or spatial uncertainty (Guo and Murphy, 2012). In this study, the uncertainty mainly lays with the emission factors that the tools use, as the input data will be provided by the user based on records or measurements (Colomb et al., 2012). Even where the IPCC calculation methodology has provided an emission range of 0.3–3% kg N_2O–N/kg N applied for artificial N application to soil (De Klein et al., 2006), most tools appear to select the default of 1%. It is suggested that tools that adopt higher Tier IPCC approaches, such as the Cool Farm and RSB Tools, reduce the uncertainty in

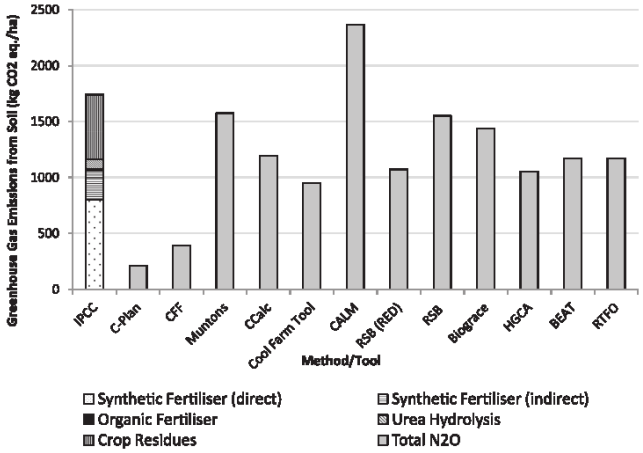

Fig. 3. Expected and observed GHG emissions from soil due to fertiliser application (based on data from Table 3).

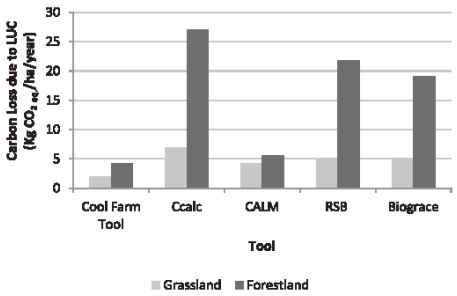

Fig. 4. Variation in estimates for LUC from grassland and forestland to arable land.

their results compared to using IPCC defaults (Guo et al., 2011), yet the actual uncertainty is not presented. Providing a single result, rather than a range or level of uncertainty, is limited in that it will only give users an indication of the average GHG emission for their crop, or a baseline from which mitigation projects can be compared.

GHG emissions from farm machinery are some minor sources of emissions that are included in some tools and not others. The CFF tool attributes the total GHG emissions from manufacture of farm machinery to the year in which the user provides information for; therefore these are overestimated. This is a consequence of the tool accounting for the farm-level emissions over an unspecified period of time, rather than the emissions for one year's work. In this study, the emissions for farm machinery construction were readjusted so they are allocated temporally to the time required to cultivate and harvest one hectare of wheat (Table 3) and it is assumed that the machinery has a working lifetime of 7000 h (EcoInvent, 2007). The guidelines for the CFF Tool state that after 10 years an item has 'paid off its carbon debt' meaning that the emission from manufacture do not need to be included, therefore farmers should use older machinery when possible (CFF, 2009). This logic is somewhat misguided, as although maximising the use of life of an item will reduce the relative emissions for 1 h's work, older machinery is less likely to be fuel efficient and conform to current emission standards. When allocated correctly to the crop level the contribution of farm machinery construction is low, as is also found in BEAT and the RSB (Fig. 2). The RSB also includes this source of emissions, though not when applying the RED methodology (RSB, 2012), as this specifically excludes emissions from building and machinery construction.

The emissions from animal manure are usually minor, depending on the quantities used. Four tools (C-Plan, CFF, CCalc and BEAT) included the emissions from manure delivery, though the majority excluded this. These may be excluded from the calculations as they are a relatively minor source of emissions, compared to artificial fertilisers. Four tools (HGCA, CALM, RSB and Cool Farm) account for N_2O emissions due to manure application, whereas these are apparently excluded from the others, though there is generally insufficient transparency in which to determine this fully. Some justification of this exclusion is that the N_2O emissions are attributed to the meat sector, in the waste disposal phases of animal husbandry. The tools that exclude emissions from manure may assume that the emissions are accounted for in the meat sector. The IPCC calculation methodology specifies, however, that all manure and organic fertiliser application should be attributed to the crop that receives it for fertilisation purposes; therefore this should be included in all tools.

In summary, the comprehensiveness of the bioenergy tools is better scoring than the farm tools and shows a higher level of homogeneity in the results. One possible explanation for this is that there are currently some existing GHG reporting methodologies in the biofuel sector, and biofuel producers are obliged to report on their emissions. The PAS2050 methodology is applicable to the food and agricultural sectors but it is currently voluntary. The RED policy in the biofuel sector differs to the PAS2050 at the field-level (Whittaker et al., 2011). The methodology outlined in the RED underlies the calculation methodology in the Biograce, RSB and RFA Tools. The comprehensive score for the RFA Tool is comparatively lower as it does not differentiate between fertiliser types, lacks N_2O from crop residues and uses IPCC Tier 1 emission factors. The HGCA and BEAT Tools were developed before the RED or RTFO were established and achieved lower comprehensiveness scores. Therefore there is evidence that since then, GHG accounting methodologies and awareness of sustainability issues has aided the development of tools that provide a more comprehensive account of GHG emissions from cultivation.

3.4. Transparency and emission factors

Transparency was assessed according to the accessibility of the inbuilt emission factors and calculation methodology, therefore this is somewhat linked to comprehensiveness. Transparency provides the user to identify the data sources used and calculation methodologies followed so that their quality or relevance can be assessed or scrutinised. Some tools provide details of the calculations, whereas some could be described as 'black boxes' where only the inputs and outputs are visible (Carvalho et al., 2012). Overall, the bioenergy-based tools achieved a higher transparency score (67%) than the farm-based tools (49%). This may also be a result of the goal and scope of the two types of tools: farm-based tools are more likely to be used by non-LCA practitioners, whereas bioenergy-based tools are designed to be used by those with expertise in LCA.

A recently published study compared the GHG emissions calculated by the Biograce and RSB Tools for a number of biofuel supply chains and showed that they produce different results despite them both complying with the RED methodology (Hennecke et al., 2012). The differences are due to different emission factors used for fertiliser manufacture and N_2O emissions from soil, and estimates for LUC differ across tools, particularly for conversion of forestland (Hennecke et al., 2012). A similar result is found here (Fig. 5). The sources of emission factors for fertiliser manufacture differ across tools (Table 5); therefore the overall result is highly sensitive to the data sources used. Only 4 of the 11 tools provided separate emission factors for ammonium nitrate and urea, despite that emissions from ammonium nitrate are approximately 67% higher than urea (Brentrup and Palliere, 2008). The importance of harmonised emission factors in GHG calculation highlights a policy and methodological gap that should be addressed in future tools and methodologies (Hennecke et al., 2012).

As mentioned in Section 3.3, the estimated N_2O emissions from soil vary between the tools (Fig. 3); and this is also a major cause of variation in the results (Fig. 5). The general lack of transparency means that it is difficult to determine why this occurs. Many of the tools state that they follow IPCC Tier 1 emission factors, and few have applied Tiers 2 or 3, though the details of the calculations are not transparent. It is suggested that a Tier 3 approaches are more appropriate for accurately assessing N_2O emissions from a particular site (Whitaker et al., 2010), and this is adopted by both the Cool Farm and RSB Tools. This involves utilising modelled emission factors that are specific to the country and are validated through

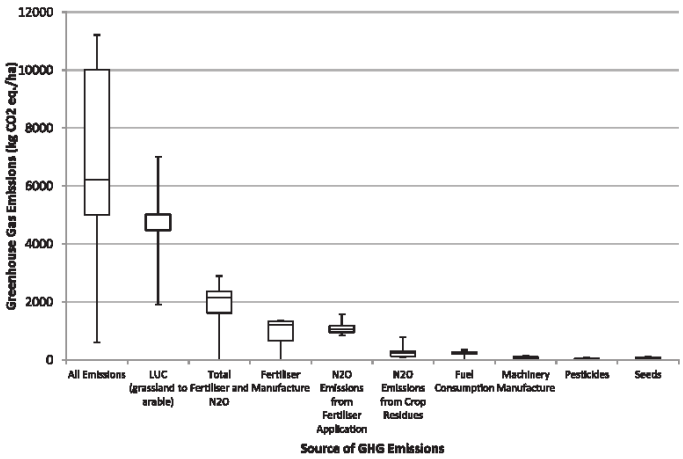

Fig. 5. A quartile box-plot showing sources of variation in the overall emissions for wheat cultivation.

experimental measurements (De Klein et al., 2006). The Cool Farm Tool utilises emission factors on specific soil parameters and on specific fertiliser types (Hillier et al., 2011, 2012). The RSB calculations are based on Ecoinvent (Nemecek et al., 2007) when following the RSB methodology, or the IPCC (Tier 2) when following the RED methodology (De Klein et al., 2006). Fig. 3 demonstrates that although a Tier 3 approach is adopted, there is still variation in the calculated emissions. Therefore some validation of the most appropriate IPCC Tier 3 emission factors may be required for purposes of harmonisation, and it is expected that these emission factors will depend on location. It is expected that, by 2014, a series of UK regional maps of local N_2O emission factors will be developed (Whitaker et al., 2010) which may facilitate a more consistent approach between tools.

There is also a lack of transparency with regards to N_2O emissions from crop residues also contribute to the overall GHG emissions from wheat cultivation (Fig. 3). These were clearly included in 6 of the 11 tools and the magnitude ranged between 96 and 782 kg CO_2 eq./ha, or between 5% and 33%. The RSB Tool includes this source of emissions but does not report separate information on this. Some tools require the user to specify the fate of straw, i.e. how much was incorporated or removed. Where the yield of straw is not specified by the user, the tools may estimate this using the IPCC (Table 11.2, De Klein et al., 2006). BEAT assumes that straw is removed from the site, and the emission estimate for crop residue incorporation represents stubble incorporation. Differences in this aspect of the calculation may be a cause of the variation of the results.

In summary, transparency varies across tools and there is evidence that the majority of the variation in GHG emissions is caused by fertiliser manufacture and N_2O emissions, which represent over 60% of total emissions in those tools that include them (Fig. 5). There is evidence that the estimates for GHG emissions from fertiliser manufacture vary because the tools use different references (Table 5). Estimates from LUC can also range considerably, particularly for forestland conversion; but this is excluded in some tools. The variation in crop residues is due to the assumptions made on the yield of straw being ploughed into the soil. GHG Emissions from fuel consumption vary little across tools. There is also minor variation in the final results due to machinery manufacture, pesticides and seeds.

3.5. Implications of results

This study has demonstrated that a number of tools available in the public domain have the ability to calculate the GHG emissions from the same process, yet can provide very different results. Some of the selected tools appear to either exclude or incompletely

Table 5
Emission factor estimates for fertiliser manufacture across tools.

Fertiliser	Emission factor per kg nutrient (kg CO_2 eq./kg)											
Tool	C-Plan	CFF	Muntons	Cool farm	CALM	CCalc	RSB	RSB (RED)	Biograce	HGCA	BEAT	RTFO
Reference cited	None	No specific reference	None	EcoInvent, 2007	No specific reference	North Energy Associates 2006	Ecoinvent	Biograce	(EUCAR et al., 2006)	HGCA, 2011	North Energy Associates 2006	No specific reference
'N fertiliser'	0.63[ac]	–	9.21[bc]	–	–	6.98	–		5.88	6.69	6.92	5.92
Ammonium nitrate (N)	–	–	–	6.20	3.80	–	8.55	8.16	–	–	–	–
Urea (N)	–	–	–	1.48	1.24	–	3.30	3.07	–	–	–	–
Phosphorous (P_2O_5)	–	–	2.2[c]	1.3	–	1.86	2.02	1.73	1	0.71	1.85	1.01
Potassium (K_2O)	–	–	0.5[c]	1.5	–	1.77	1.44	1.12	0.6	0.46	1.76	0.58

This is deduced rather than stated in the tool or supporting manual.
[a] This is per kg of fertiliser.
[b] Includes N_2O emissions from soil.

calculate some major sources of GHG emissions, particularly LUC and N_2O emissions from soil, therefore their credibility is questionable (Whitaker et al., 2010).

One potential implication of this is that some tools may be misused and used to calculate GHG emissions for activities that they are not designed for. For example, farm-based tools that are designed to calculate GHG emissions on a farm level will require information on various aspects of farm management that may exceed what should be attributed to a specific crop. It can become difficult to attribute GHG emissions farm activities to the production of a given 'tonne of wheat'. As this is not the original goal and scope of the tool, they should not be used in this way. An exception is the Cool Farm Tool, which can be used to calculate the GHG balance of a single crop, or a single crop farm, and is one of the most highly rated tools observed in this study. It is possible that other tools that are not in the public domain, or available in the future, could also be suitable for GHG accounting in agriculture.

Using a bioenergy tool to estimate the GHG emissions from a given crop may be more difficult to perform by a non-LCA practitioner, as these tools are generally less user friendly. Bioenergy tools are designed to assess the GHG emission savings of a biofuel against strict targets. Hennecke et al. (2012) highlights that if biofuel producers have an option of tools to calculate their GHG emissions, then it is likely they may select a tool that generates the greatest GHG savings. Even when the tool complies with the RED, differences in the emission factors may yield different results.

4. Conclusion

"All models are wrong, but some are useful" (Box, 1976). As models are built from a combination of methodology, data and informed assumptions, any differences will naturally give different results, but sometimes there are no right or wrong answers, just a reflection of a different perspective on how to calculate the emissions (Aylott et al., 2011, 2012). Both the farm and bioenergy tools can be used to calculate the same thing (i.e. one hectare of wheat); yet after comparing some tools that are available in the public domain, they appear to produce very different results.

GHG calculation models that follow a LCA approach require careful planning so that the model is fit to fulfil its original goal and scope. A tool designed to provide a comprehensive account of the GHG emissions from crop production for food or for biofuel production should include emissions from soil and fertiliser manufacture in the system boundaries, as these represent approximately 80% of total emissions. Tools excluding these from the system boundaries will not provide a full account of the emissions resulting from arable cropping, therefore they should be avoided. LUC is also another important source of GHG emissions that should be included in any GHG calculator that utilises land.

Suitable GHG calculator tools include those that present both a user-friendly platform for use, and provide a comprehensive account of GHG emissions occurring on a farm or product level. This study identified the Cool Farm Tool as the most highest-rated tool that is available in the public domain and free to use. This is recommended for use for single crop assessments. For whole-farm assessments the CALM Tool is recommended. For biofuel assessments that are RED-compliant, the RSB Tool or Biograce are recommended.

In terms of the actual GHG estimates, emissions from LUC are the largest sources of variation, but this was not featured in all tools. After this, N_2O emissions from soil and fertiliser manufacture are also significant sources of variation between the results calculated. As transparency is lacking, it is difficult to determine whether this variation is a result of the system boundaries or the calculation methodologies employed. The variation in fertiliser manufacture is due to a combination of data sources used by each tool. As fertiliser manufacture is typically a major contributor to total GHG emissions from an arable crop, small differences in the emission factor can generate very different results.

This study found that farm-based tools are more user-friendly than bioenergy-based tools; though the latter achieved a higher rating in the level of information provided in the results, the transparency of the underlying assumptions and the comprehensiveness of emission sources included in the calculations. An exception to this is the Cool Farm Tool, which was the highest rated. The differences in scoring between the selected farm and bioenergy tools may be due to their original goal and scope: farm-based tools are generally designed to inform users on the sources and mitigation options on a farm level; bioenergy-based tools provide information on the GHG emissions from producing a single crop. The differing goal and scope of the two approaches may affect the design of the subsequent GHG calculation tool, and hence the results.

This study has demonstrated that different goal and scopes, system boundaries and underlying emission factor data within GHG calculation tools can result in very different results despite the same input data used. The goal and scope of a tool is the most important factor in determining its intended use. Bioenergy-based tools demonstrated less variation across the results than farm-based tools, which may be due to the methodological guidelines available for biofuel reporting. Therefore, there is a need to harmonise both methodology and emission factors in biofuel GHG calculation tools, so that at least a consistent result can be generated.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envsoft.2013.03.015.

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13.3. Appendix 3: Emission Factors

Description	Unit	GHG Emissions (kg CO_2 eq.unit^{-1})				Reference
		CO_2	CH_4	N_2O	Total	
Diesel fuel	MJ	0.105	0.003	0.001	0.109	(EcoInvent, 2007)
Electricity	MJ	0.144	0.000	0.001	0.145	(DEFRA and DECC, 2012)
LPG	MJ	0.070	0.001	0.000	0.071	(AEA Technology and North Energy Associates, 2010)
Natural Gas	MJ	0.054	0.003	0.000	0.057	
Gasoline	MJ				0.838	(Biograce.net, 2012)
Transport	t-km	0.005	0.000	0.000	0.005	(EcoInvent, 2007)
Pesticides	kg a.i	8.716	0.779	0.559	10.075	
Seed	kg	0.214	0.009	0.353	0.576	
Super 3x Phoshpate	kg P_2O_5	1.926	0.079	0.013	2.021	
Muriate of Postash	kg K_2O	0.442	0.044	0.011	0.497	
Ammonium Nitrate	kg N	2.760	0.150	5.639	8.551	
Amm. Nitrate (BAT)	kg N	1.987	0.148	0.850	2.987	
Urea	kg N	3.089	0.201	0.012	3.304	
Amm. sulphate	kg N	2.544	0.136	0.009	2.691	
Compost	kg	4.146	0.335	0.025	4.524	
Delivered Manure	tonne	0.011	0.000	0.000	0.011	
Heavy Ploughing	hour	0.811	0.053	0.004	0.869	(EcoInvent, 2007) Weight from (Williams et al., 2006)
Power Harrowing	hour	1.323	0.086	0.006	1.416	
Subsoiling	hour	1.141	0.074	0.005	1.221	
Heavy Discing	hour	2.693	0.176	0.012	2.882	
Light Discing	hour	2.646	0.172	0.012	2.832	
Drilling	hour	1.058	0.069	0.005	1.133	
Combined Drill	hour	2.693	0.176	0.012	2.882	
Rolling	hour	1.870	0.122	0.008	2.001	
Sprayer	hour	0.507	0.034	0.002	0.544	
Fertiliser	hour	0.101	0.007	0.000	0.109	
Combine harvester	hour	6.812	0.550	0.041	7.432	
Baler	hour	1.058	0.069	0.005	1.133	
Tractor	hour	2.316	0.151	0.010	2.480	
Mower	hour	0.412	0.027	0.002	0.441	
Miscanthus Planter	hour	1.352	0.088	0.006	1.448	(EcoInvent, 2007) Weight from (Ercoli et al., 1999)
Forage Harvester	hour	4.608	0.372	0.028	5.028	
Caustic Soda (49%)	kg	1.040	0.047	0.010	1.096	(EcoInvent, 2007)
DiammoniumPhosp 21%)	kg	0.316	0.012	0.002	0.330	
Sulphuric Acid 93%)	kg	0.119	0.004	0.001	0.124	
Calcium Chloride	kg	0.798	0.050	0.004	0.853	
Yeast	kg	0.973	0.045	0.016	1.035	
Lime	kg	0.002	0.000	0.000	0.002	
Nutrient Feed	kg	0.233	0.166	0.014	0.413	
Enzyme AMG	kg	2.250	0.000	0.000	2.250	(AEA Technology and North Energy Associates, 2010)
Enzyme α amylase	kg	1.600	0.000	0.000	1.600	
Sulphur Dioxide	kg	0.130	0.101	0.204	0.435	
Sodium Hydroxide	kg	1.120	0.004	0.008	1.132	

13.4. Appendix 4: Parameters for N_2O Emissions

The following table shows the parameters used to calculate direct and indirect N_2O emissions from artificial, organic and crop residue addition to soil (De Klein and Van Logtestijn, 1996).

Parameter	Description	Crop	Best Guess	S.D or Range	Distribution
Area	Total annual harvested area (ha timeframe^{-1})	Both	1	n/a	n/a
$Area_{burnt}$	Total annual area of crop T burnt (ha timeframe^{-1})	Both	0		
C_f	Combustion factor (dimensionless)	Wheat	0.9		
		Miscanthus	0.8[$]		
$Frac_{renew}$	Fraction of crop T that is renewed annually	Wheat	1		
		Miscanthus		0.05 - 0.2	
Slope	From table 11.6 in De Klein et al. (2006)	Wheat	1.61	0.02	Normal
		Miscanthus	0.3	0.08	
Intercept		Wheat	0.4	0.05	Normal
		Miscanthus	0	n/a	
N_{AG}	N content of above ground residues of crop T (kg N D.M^{-1})	Wheat	0.6	n/a	n/a
		Miscanthus	0.015		
$Frac_{remove}$	N content of above ground residues of crop T (kg N D.M^{-1})	Wheat	0.66	n/a	n/a
		Miscanthus	n/a	n/a	
R_{BIO}	Ratio of above to below ground residues	Wheat	0.23	0.136 - 0.324	Uniform
		Miscanthus	0.8	0.4 – 1.2	
N_{BG}	N content of below ground residues (kg N D.M^{-1})	Wheat	0.009	n/a	n/a
		Miscanthus	0.012		
EF N addition to soil *	Emission factor for applied N to soil (kg N-N_2O kg N^{-1})	Both	0.01	0.03 - 0.03	Uniform
$Frac_{leached}$ from N	Fraction of applied N that is leached	Both	0.3	0.1 - 0.8	
EF leached N *	Emission factor for leached N (kg N-N_2O kg N leached^{-1})	Both	0.0075	0.0005 – 0.025	
$Frac_{GASF}$ of N	Fraction of applied artificial N that is volatised	Both	0.1	0.03 - 0.3	
$Frac_{GASM}$ of N	Fraction of applied organic N that is volatised	Both			
EF volatised N *	Emission factor for volatised N (kg N-N_2O kg N volatised^{-1})	Both	0.01	0.002 – 0.05	

13.5. Appendix 5: Diesel Fuel Consumption for Miscanthus

The following table provides the diesel fuel estimates for Miscanthus taken from literature and from contractors.

Phase	Operation	Fuel Consumption (litres ha^{-1})			Literature (litres ha^{-1})	Notes from Contractors
		Average	Min	Max	Range	
Establishment	Ploughing	22	15	30	6[a]-58[b]	Heavy soil avoided.
	Power Harrow	70	-	-	4[c]-40[d]	Not much variation.
	Planting	30	-	-	4[e]-22[f]	Not much variation.
	Rolling	17	-	-	2[g]-10[h]	Not much variation.
	Spraying	7	-	-	1[i]-3[j]	Not much variation.
Maintenance	Mower	7	-	-	8.5[k]	Not much variation.
Harvesting Biomass	Forage Harvesting	16	-	21	10[l]-99[m]	Can increase with the thickness of the plant.
	Baling & Haulage	3.46	-	-	Depends on yield	Average for 2011 (units are litres tonne^{-1})
Rhizome Harvesting	Scalp Flail	9	-	-	-	Not much variation.
	Flat Lift	22	-	-	-	Variation is not high (not propagated on heavy soil.)
	Rotospike	29	-	-	-	
	Lifting	30	-	-	23[n]	
	Tractor and Trailer (to sorting mill)	120	-	190	-	Typically involves 7 trips with a 14 tonne trailer per hectare.
	Return of Soil/Stones	85	-	135	-	5 return trips usually required.
	Loader	22	-	-	10[o]	
	Bulldozer	7	-	-	-	
Termination	Subsoiler	22	-	-	-	
	Spray	7	-	-	1-3	Not much variation.

a. Chisel ploughing, low estimate (Lal, 2004b)

b. Estimated deep plough (Nix, 2011)

c. (Nemecek et al., 2007)

d. Estimated power harrow (Nix, 2011)

e. (Gilbert et al. 2011)

f. (Ercoli et al., 1999)

g. (Williams et al., 2006)

h. Estimated rolling (Agro Business Consultants Ltd, 2011)

i. Spraying, low estimate (Lal, 2004b)

j. (Heller et al., 2003)

k. (Smeets et al., 2009)

l. (Smeets et al., 2009)

m. (Ercoli et al., 1999) – *May include forwarding and baling as this number is high.*

n. (Hillier, 2012)

o. (Felten et al., 2013)

13.6. Appendix 6. Estimating GHG Emissions from Cellulase Manufacture

The following tables show estimates for cellulase manufacture from two key resources used in this study.

Table 13-1.Energy requirements for cellulase production (A. Borrion pers. com. 2012).

Input	Requirement (MJ tonne straw^{-1})	Emission Factor (kg CO_2 eq. MJ^{-1})	GHG emissions (kg CO_2 eq. tonne straw^{-1})
Natural gas	33.0	0.060	1.98
Diesel	0.4	0.086	0.04
Heavy fuel oil	19.8	0.090	1.79
Total			**3.80**

Table 13-2. Energy requirements for cellulase manufacture (Slade et al., 2009).

Stage	Parameter	Unit	Value
Fermentation	Agitation Power Requirement	W m$^{3-1}$	400
	Air Sparge Power Requirement	W m$^{3-1}$	2183
	Residence Time	H	160
	Total Fermentor Volume	m^3	1000
	Active Volume of Fermentor	%	80
	Total Electricity Input	MJ fermentor^{-1}	7439
	Capacity	litres	800,000
Yield	Cellulase Output	FPU litre^{-1} hour^{-1}	75.0
		FPU fermentor^{-1}	9.6 x 10^9
Demands	Electricity Required for Cellulase	MJ FPU^{-1}	7.7 x 10^{-7}
	Cellulase for Conversion	FPU tonne straw^{-1}	4.9 x 10^6
	Emission Factor for Electricity	kg CO_2 eq. MJ^{-1}	0.583
Total	**GHG Emissions for Cellulase**	**kg CO_2eq. tonne straw^{-1}**	**2.21**

13.7. Appendix 7. Estimating the Probability Density Function

To identify the probability density function of each parameter, the following methods were applied. For parameters with few data points, a uniform distribution was assumed between minimum and maximum points. Where the dataset was larger, i.e. from the farmer questionnaires, the distribution of the data must be identified.

The methodology for identifying the most appropriate distribution is similar to that followed by Guo & Murphy (2012). Firstly, it is determined whether or not the dataset is skewed and does or does not follow a normal distribution. To do this, the moment coefficient of skewness and kurtosis is calculated. The equation identifies, with a 95% confidence level, whether or not the data is skewed. If the dataset is skewed then it is assumed that it cannot be normally distributed, and a visual estimation was made using the Distribution Fitting Tool in MATLAB, to establish a suitably fitting distribution. Symmetrical datasets were identified as normally distributed if they are not significantly different to a simulated normal distribution, via a Pearson's Chi Squared Test (**Equation A 5**)

A Pearson's Chi Squared Test involves calculating *gi* (the coefficient of skewness and kurtosis (**Equation A 1**)), SES (standard error of skewness (**Equation A 2**)), *G1* (the sample skewness (**Equation A 3**)), and the *Zgi* (test statistic (**Equation A 4**)). A *Zgi* value of greater or less than 2/-2 indicates that the data is significantly skewed with a 95% confidence level (Brown, 2013). Symmetrical datasets were identified as normally distributed if they are not significantly different to a simulated normal distribution, via a Pearson's Chi Squared Test. If the dataset is skewed then it cannot be normally distributed, and a visual estimation was made using the Distribution Fitting Tool in MATLAB, to establish whether the dataset is log-normally, triangularly or uniformally distributed.

Equation A 1. Calculating *gi*, the coefficient of skewness and kurtosis.

$$gi = {m_3}\Big/{m_2^{\frac{2}{3}}}$$

$$\textit{Where: } m_3 = \frac{\sum(x-\bar{x})^3}{n} \quad \textit{and} \quad m_2 = \frac{\sum(x-\bar{x})^2}{n}$$

Equation A 2. Calculating SES, the standard error of skewness.

$$SES = \sqrt{\frac{6n\,(n-1)}{(n-2)(n+1)(n+3)}}$$

Equation A 3. Calculating *G1*, the sample skewness.

$$G_1 = \frac{\sqrt{n(n-1)}}{n-2}\ gi$$

Equation A 4. Calculating *Zgi*, the test statistic.

$$Zgi = {}^{G_1}/_{SES}$$

Equation A 5. Pearson's Chi Squared Test.

$$X^2 = \sum_{i=1}^{n} \frac{(O_i - E_i)^2}{E_i}$$

Where:

X^2 = Pearson's cumulative test statistics

O_i = Observed frequency

E_i = Expected frequency

n = number of pairs

A dataset is significantly different to a normal distribution if X^2 is greater than the degrees of freedom.

13.8. Appendix References

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